

MANAGING FOREST ECOSYSTEMS

Developing Principles and Models for Sustainable Forestry in Sweden

Edited by

Harald Sverdrup and Ingrid Stjernquist



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DEVELOPING PRINCIPLES AND MODELS FOR SUSTAINABLE
FORESTRY IN SWEDEN

Managing Forest Ecosystems

Volume 5

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Aims & Scope:

Well-managed forests and woodlands are a renewable resource, producing essential raw material with minimum waste and energy use. Rich in habitat and species diversity, forests may contribute to increased ecosystem stability. They can absorb the effects of unwanted deposition and other disturbances and protect neighbouring ecosystems by maintaining stable nutrient and energy cycles and by preventing soil degradation and erosion. They provide much-needed recreation and their continued existence contributes to stabilizing rural communities.

Forests are managed for timber production and species, habitat and process conservation. A subtle shift from *multiple-use management* to *ecosystems management* is being observed and the new ecological perspective of *multi-functional forest management* is based on the principles of ecosystem diversity, stability and elasticity, and the dynamic equilibrium of primary and secondary production.

Making full use of new technology is one of the challenges facing forest management today. Resource information must be obtained with a limited budget. This requires better timing of resource assessment activities and improved use of multiple data sources. Sound ecosystems management, like any other management activity, relies on effective forecasting and operational control.

The aim of the book series *Managing Forest Ecosystems* is to present state-of-the-art research results relating to the practice of forest management. Contributions are solicited from prominent authors. Each reference book, monograph or proceedings volume will be focused to deal with a specific context. Typical issues of the series are: resource assessment techniques, evaluating sustainability for even-aged and uneven-aged forests, multi-objective management, predicting forest development, optimizing forest management, biodiversity management and monitoring, risk assessment and economic analysis.

The titles published in this series are listed at the end of this volume.

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Contents

| | | |
|----------|--|-----------|
| 1 | Introduction to the SUFOR Programme | 1 |
| 1.1 | Foreword by the editors | 1 |
| 1.2 | Foreword by the programme director | 4 |
| 1.3 | SUFOR Programme formulation | 5 |
| 1.3.1 | Problem description | 5 |
| 1.3.2 | Hypotheses of the programme | 10 |
| 2 | On Swedish forests | 13 |
| 2.1 | The forests in southern Sweden | 13 |
| 2.1.1 | Forest characteristics | 13 |
| 2.1.2 | SUFOR experimental sites | 16 |
| 2.2 | Forest ownership in Sweden | 16 |
| 2.2.1 | Ownership structure | 16 |
| 2.2.2 | Private owners and ecological sustainability | 18 |
| 3 | Defining sustainability | 21 |
| 3.1 | Introduction | 21 |
| 3.2 | The dimensions of sustainability | 21 |
| 3.3 | Aspects of sustainability | 23 |
| 3.3.1 | Natural sustainability | 23 |
| 3.3.2 | Social sustainability | 25 |
| 3.3.3 | Economic sustainability | 26 |
| 3.3.4 | Temporal dimensions | 28 |
| 3.3.5 | The spatial dimensions | 29 |
| 3.3.6 | Biodiversity | 30 |
| 3.3.7 | Harvest sustainability | 30 |
| 3.3.8 | The dynamics of sustainability | 30 |
| 3.4 | Integrated sustainability | 31 |
| 4 | Principles of sustainable forest management | 33 |
| 4.1 | Introduction | 33 |
| 4.2 | Mixed-species stands as a component | 35 |
| 4.3 | Landscape and forests | 36 |
| 4.4 | A time of changes | 37 |
| 4.4.1 | Global climate | 37 |
| 4.4.2 | Pollution climate | 37 |
| 4.4.3 | The market | 38 |
| 4.5 | A future market outlook | 38 |
| 4.5.1 | Paper and pulp market | 39 |
| 4.5.2 | Sawmill products | 40 |
| 4.5.3 | Wood for energy | 41 |
| 4.6 | Certification | 42 |

| | | |
|----------|--|-----------|
| 4.6.1 | Introduction | 42 |
| 4.6.2 | Certified management | 42 |
| 4.6.3 | Forest Stewardship Councils; FSC and PEFC | 42 |
| 4.6.4 | Conclusion | 43 |
| 4.7 | The management planning procedure | 43 |
| 4.7.1 | Introduction | 43 |
| 4.7.2 | Planning structure | 45 |
| 4.7.3 | Economic principles | 48 |
| 4.7.4 | The problem of prices and interest rates | 53 |
| 4.7.5 | Risks | 54 |
| 4.7.6 | Future development | 55 |
| 4.8 | Concluding remarks | 56 |
| 5 | Integrated modelling | 57 |
| 5.1 | On the principle of modelling in general | 57 |
| 5.2 | Modelling the forest system | 62 |
| 5.2.1 | Introduction | 62 |
| 5.2.2 | Forest model history | 62 |
| 5.2.3 | The space/time realm of forest ecosystem modelling | 63 |
| 5.2.4 | Model roles | 64 |
| 5.2.5 | Scaling | 65 |
| 5.2.6 | Different model approaches | 66 |
| 5.2.7 | Overview of selected models | 68 |
| 5.3 | The Integrated ForSAFE model | 74 |
| 5.3.1 | Finding a forest growth model | 76 |
| 5.3.2 | Merging PnET, SAFE and DECOMP | 77 |
| 5.3.3 | The SAFE model and soil chemistry | 83 |
| 5.4 | Summary | 83 |
| 6 | Models of the risk of windthrow and frost | 85 |
| 6.1 | Introduction | 85 |
| 6.2 | The risk of windthrow | 85 |
| 6.2.1 | Model | 86 |
| 6.2.2 | Evaluations | 87 |
| 6.2.3 | Discussion | 88 |
| 6.3 | The relative risk of frost | 88 |
| 6.3.1 | Model | 89 |
| 6.3.2 | Results | 90 |
| 6.3.3 | Discussion | 90 |
| 6.3.4 | Acknowledgements | 90 |
| 7 | Biogeochemical processes and mechanisms | 91 |
| 7.1 | Introduction | 91 |
| 7.2 | Aspects of growth and nutrient uptake | 92 |
| 7.2.1 | Plant growth and nutrient uptake regulation | 92 |
| 7.2.2 | Nitrogen tree uptake processes | 95 |
| 7.2.3 | Base cation uptake processes | 100 |
| 7.2.4 | Modelling forest growth | 107 |
| 7.2.5 | Allocation and growth | 109 |
| 7.3 | Decomposition of organic matter | 114 |
| 7.3.1 | Introduction | 114 |
| 7.3.2 | Objectives | 115 |
| 7.3.3 | Model concept | 116 |

| | | |
|----------|--|------------|
| 7.3.4 | Data and Methods | 118 |
| 7.3.5 | Results | 119 |
| 7.3.6 | Discussion of data evaluation | 134 |
| 7.3.7 | DECOMP; Integrated decomposition modelling | 134 |
| 7.4 | Nitrogen Immobilization | 146 |
| 7.4.1 | Introduction | 146 |
| 7.4.2 | Theory | 146 |
| 7.4.3 | Results | 147 |
| 7.5 | Nitrification | 148 |
| 7.5.1 | Nitrification processes | 148 |
| 7.5.2 | Theory | 148 |
| 7.5.3 | Nitrification dependence on temperature | 151 |
| 7.5.4 | Discussion | 152 |
| 7.6 | Denitrification | 153 |
| 7.6.1 | Theory | 153 |
| 7.6.2 | Data | 154 |
| 7.6.3 | Results | 155 |
| 7.6.4 | Discussion | 159 |
| 7.6.5 | Conclusion | 162 |
| 7.7 | The SAFE model | 163 |
| 7.7.1 | Mass balance equations | 163 |
| 7.7.2 | Soil solution chemistry | 165 |
| 7.7.3 | Cation exchange | 166 |
| 7.8 | Weathering of soil minerals | 172 |
| 7.8.1 | Weathering and the PROFILE model | 173 |
| 7.8.2 | Calculating field weathering rates | 183 |
| 7.8.3 | Estimating soil mineralogy inputs to the model | 183 |
| 7.8.4 | Including clay minerals in the model | 186 |
| 7.8.5 | Some persistent myths in weathering | 191 |
| 8 | Forest vitality and stress implications | 197 |
| 8.1 | Mechanisms and systems of multiple stress | 197 |
| 8.1.1 | Interaction of stress factors | 197 |
| 8.1.2 | Memory effects | 200 |
| 8.1.3 | Forests management as a stress factor | 200 |
| 8.1.4 | Plant strategy and stress | 201 |
| 8.1.5 | System analysis of the stress/health relationship | 203 |
| 8.2 | Forest health indicators | 204 |
| 8.2.1 | Visible indicators for assessing tree vitality | 204 |
| 8.2.2 | Nutrient concentrations and ratios to N as vitality indicators | 208 |
| 8.2.3 | Soil chemistry | 210 |
| 8.2.4 | Pathogens and insects as vitality indicators | 211 |
| 8.2.5 | Evaluation of vitality indicators for trees and ecosystems | 212 |
| 8.3 | Sensitivity to frost and drought | 213 |
| 8.3.1 | Frost | 213 |
| 8.3.2 | Drought | 214 |
| 8.4 | Air pollution effects | 216 |
| 8.4.1 | Ground-level ozone | 216 |
| 8.4.2 | Acid deposition and soil acidity | 222 |
| 8.4.3 | Experimental results | 230 |
| 8.4.4 | Discussion | 235 |
| 8.4.5 | Nitrogen and nutrient imbalance | 236 |
| 8.5 | Stress by pathogens and pests | 245 |

| | | |
|-----------|---|------------|
| 8.5.1 | Pest and pathogens in Swedish forests | 245 |
| 8.5.2 | Effects of nutrient imbalances on pest and pathogens | 246 |
| 8.6 | Countermeasures against stress symptoms | 249 |
| 8.6.1 | Restrictions of air pollutants | 250 |
| 8.6.2 | Adaptation of silviculture | 250 |
| 8.6.3 | Application of lime, wood-ash and mineral fertilizers | 253 |
| 8.7 | Forest conditions — A regional overview | 256 |
| 8.7.1 | Background | 256 |
| 8.7.2 | Regional co-operation for assessment of decline and stress | 257 |
| 8.7.3 | Regional forest decline surveys | 257 |
| 8.7.4 | Surveys of foliage chemistry | 262 |
| 8.8 | Estimation of total risk for forest health | 270 |
| 9 | Biodiversity in sustainable forestry | 273 |
| 9.1 | Introduction | 273 |
| 9.2 | The Swedish cultural landscape and biodiversity | 275 |
| 9.3 | The need for system analysis of biodiversity | 277 |
| 9.4 | Modelling strategy in SUFOR | 279 |
| 9.4.1 | Ground vegetation modelling | 280 |
| 9.4.2 | Biodiversity of insects | 281 |
| 9.5 | Conclusions | 281 |
| 9.6 | Principles of biodiversity preservation in forestry | 284 |
| 9.6.1 | Introduction | 284 |
| 9.6.2 | Definition of sustainable biodiversity | 285 |
| 9.6.3 | Establishing the baseline of forest biodiversity | 285 |
| 9.6.4 | Structure and biodiversity of the natural forests of Europe | 286 |
| 9.6.5 | Human impact in the forests of Europe | 287 |
| 9.6.6 | Differences between southern Sweden and northern Europe | 287 |
| 9.6.7 | Palaeontomological studies of biodiversity in Sweden | 288 |
| 9.6.8 | Disturbance regimes of southern Sweden | 288 |
| 9.6.9 | Critical features for biodiversity preservation in forests | 291 |
| 9.6.10 | Recommendations for biodiversity restoration | 295 |
| 9.6.11 | Conclusions for a sustainable forestry | 296 |
| 10 | Forests, Acidification and the Socio-economic Cost | 297 |
| 10.1 | Introduction | 297 |
| 10.2 | Ecology and Economics | 298 |
| 10.2.1 | Accounting the Environment | 299 |
| 10.2.2 | Socio-economic justification | 301 |
| 10.2.3 | Impact of the Discount Rate | 302 |
| 10.2.4 | Estimating the Economic Value | 302 |
| 10.2.5 | Methods for Valuing Natural Resources | 303 |
| 10.3 | Environmental Accounting in Sweden | 304 |
| 10.3.1 | Estimating the value of utilities affected by acidification | 306 |
| 10.4 | Estimating the economically acceptable loss of value | 307 |
| 10.5 | Area Affected by Acidification | 308 |
| 10.6 | Discussion | 310 |
| 10.7 | Reflection on the treat from acidification | 311 |

| | |
|---|------------|
| 11 Assessing effects of wildlife on forestry | 317 |
| 11.1 Browsing damage to forests vs. recreational values | 317 |
| 11.1.1 Introduction | 317 |
| 11.1.2 Browsing damages | 317 |
| 11.1.3 Influence on forage availability | 319 |
| 11.1.4 Recreational value | 321 |
| 11.1.5 Conclusions | 322 |
| 11.2 Dynamic modeling of browsing damages by moose | 323 |
| 11.2.1 Introduction | 323 |
| 11.2.2 Diet selection | 325 |
| 11.2.3 Conceptual modeling of forage selection | 327 |
| 11.2.4 Modelling of intake rate | 327 |
| 11.2.5 Modeling of browse utilisation on national scale | 330 |
| 11.2.6 Predicting moose population and degree of utilisation | 332 |
| 11.2.7 Conclusion | 335 |
| 12 Sustainability in spruce and mixed-species stands | 337 |
| 12.1 Introduction | 337 |
| 12.2 Objectives | 337 |
| 12.3 Site description | 338 |
| 12.4 Management description | 338 |
| 12.5 Experimental setup and sampling | 338 |
| 12.6 Chemical composition and nutrient dynamics | 340 |
| 12.7 Methods and models | 340 |
| 12.8 Biogeochemical mapping of a single stand | 341 |
| 12.9 Assessment of nutrient sustainability | 346 |
| 12.10 Assessment of economic sustainability | 349 |
| 12.11 Conclusions | 352 |
| 13 Productivity scenarios for the Asa Forest Park | 355 |
| 13.1 Asa site description | 355 |
| 13.2 Introduction | 357 |
| 13.3 The landscape model | 358 |
| 13.3.1 The tree cover model | 358 |
| 13.3.2 Nutrient budgets and nitrogen leaching | 362 |
| 13.4 Indicators | 365 |
| 13.5 Forest management | 366 |
| 13.5.1 Stand management programs | 366 |
| 13.5.2 Landscape management approaches | 369 |
| 13.6 Evaluation of the management programs | 371 |
| 13.6.1 Effects on biodiversity on land | 372 |
| 13.6.2 Nutrient budgets | 373 |
| 13.6.3 Effects on water quality | 375 |
| 13.6.4 Effects on tree vitality | 376 |
| 13.6.5 Economic effects | 377 |
| 13.7 Conclusions | 378 |
| 14 Assessment of sustainability in the Asa Forest Park | 381 |
| 14.1 Assessment of the sustainability of mineral nutrient use | 381 |
| 14.1.1 Introduction | 381 |
| 14.1.2 Data collection | 381 |
| 14.1.3 Soil sampling | 382 |
| 14.1.4 Calculation flowchart | 383 |

| | | |
|-----------|---|------------|
| 14.1.5 | Measured soil parameters and analytical methods | 384 |
| 14.1.6 | Results | 387 |
| 14.1.7 | Biogeochemical mass balances at Asa | 398 |
| 14.2 | Sustainable economic profits | 416 |
| 14.2.1 | Introduction | 416 |
| 14.2.2 | Economic model | 416 |
| 14.2.3 | Results | 416 |
| 14.2.4 | Conclusions | 417 |
| 14.3 | Approximate carbon balances | 419 |
| 14.3.1 | Introduction | 419 |
| 14.3.2 | Methods | 420 |
| 14.3.3 | Input data | 421 |
| 14.3.4 | Calculations and results | 422 |
| 14.3.5 | Discussion | 424 |
| 14.3.6 | Conclusions | 426 |
| 15 | Nutrient sustainability for Swedish forests | 427 |
| 15.1 | Introduction | 427 |
| 15.2 | Input data | 427 |
| 15.3 | Results | 428 |
| 15.4 | Discussion | 429 |
| 15.5 | Conclusions | 431 |
| 16 | General conclusions | 433 |
| 16.1 | Statements and conclusions | 433 |
| 16.2 | Risk management | 440 |
| 16.3 | The future | 441 |
| 17 | References | 443 |

Chapter 1

Introduction to the SUFOR Programme

1.1 Foreword by the editors

This report represents some of the "science footwork" performed in the research programme for Sustainable Forestry in Southern Sweden (SUFOR). SUFOR is a programme funded by MISTRA, an independent non-for profit foundation supporting strategically important environmental research in Sweden. The SUFOR programme was given 108 million Swedish kronor (15 million Euro) for eight years of research during the period from 1997 to 2005. This particular book has several purposes:



Figure 1.1: *Professor Sverdrup in front of an 800 year old Douglas Fir. Old trees sometimes provide very valuable raw material for quality sawn wood. Sustainable forest economy is favored by a balance between bulk pulpwood production and production of highly priced quality timber. Photo: Mats G. E. Svensson*

- To reflect the basic research carried out in the SUFOR programme
- To reflect how this research was integrated and put to practical use
- To reflect the possibilities that this work has created for achieving sustainable forest management

The book describes many of those small parts that patiently had to be pieced together, tested, tried, improved, in order to create the parts of a greater scheme. The initial parts of this report will draw up some of this larger scheme, the description of the single components will follow, and at the end, in the implementation part, some of it will be tested and displayed. Our work is not finished, it has not even begun, what you see is the beginning of the beginning of something that eventually in phase two of the SUFOR programme will come of age. In the SUFOR programme, the road to sustainability is built from a bottom up approach. We built it up from principles of natural sustainability, subsequently adding

economic and social principles, and in that order which is also reflected in this book. The goals and fundamental criteria may be defined from a top perspective, but because everything reaches down to the foundations of nature, this is where we start to focus. In the later part of the SUFOR programme, more elements of economic and social sustainability will be added upon the foundation of natural science principles. In the construction of the large system of model packages (models, diagrams, databases, algorithms, testing cases, scenarios, user-friendly computer programmes on CD's for forest owners, sustainability yield tables) has not yet been fully completed, they are still under construction. Such a work is long and inglorious, requiring patience with both by financial contributors as well as by the creators. Still, this may be some of the most important parts of the programme, making our results available and useful to the end user. Some components are ready for testing and will be shown in this report. The PROFILE model and it's new versions have already found wide use for weathering calculations in a large numbers of countries in all continents. The SUFOR programme has now completed its first phase according to a "learning loop" (Figure 1.3). "The learning loop" is a fundamental concept in the SUFOR Programme, it describes our systems thinking attitude and our way of iteratively learning the solution to the problems we took on.

The work in the Programme has been problem-oriented from the first beginning, the work did not start with gathering data, but by defining the problem and by formulation present knowledge as a mental model. Only after that has been documented, is data gathered, and very selectively with respect to the problem and what the defined model requires to test the hypothesis. The task of solving a problem is not finished before a full cycle has been completed, something that will take place later this year. The book may appear to have a bias towards natural science which reflects the first phase of the research programme. In the second phase, the programme has a heavier emphasis on economy and interactions with the society. The SUFOR programme has now completed its first phase according to a "learning loop" (Fig. 1.3). "The learning loop" is a fundamental concept in the SUFOR Programme, it describes our systems thinking attitude and our way of iteratively learning the solution to the problems we took on. Thus, we would ask you not to react with conservatism and defense against change, but to look beyond at all the new possibilities change may bring. The world is changing, and it does this whether we like it or not, the only thing we can decide is to adapt to the changes or not to change. The purpose of SUFOR is to prepare for a changing world, and the results from SUFOR will be focused towards how to do this. Considering the stock at risk, the probability for large scale changes and the future demand for sustainability for survival, we cannot afford not to stop, rethink and change our ways. This is not only a threat, involving risk, but also a tremendous possibility. We have it in our hands to do this, we have the knowledge base to start the development. SUFOR has taken one of these steps forward. Throughout the book, causal loop diagrams (CLD) can be found. These have been included in order to systematically analyze the dynamic properties and the behaviour of each system and subsystem. In nature, nothing is linear and feedbacks are important. Such relationships have been described by using CLD diagrams.



Figure 1.2: *Dr. Stjernquist in front of an 88 year old **Fagus sylvatica** outside Lund University. The mixing of deeper rooted trees like beech, oak or birch into stands of shallow rooted trees such as Norway spruce is an important component in increasing natural nutrient sustainability. Photo: H. Sverdrup*

These diagrams operate with a specific nomenclature. The causal loop diagrams are used to explore and map the dynamic properties of components of the forest system. Fig. 1.4 show an example for a population system. The first part consists of mature individuals (for example trees) which produce offspring proportional to their number (such as seeds). The more mature individuals (trees), the more offspring (seeds and seedlings), the more mature individuals. This part of the system will escalate, unless something limits it. The limiting system is introduced by the component of mortality. The mature individuals (trees) die (dieback and harvest) after aging in a set proportion to their number. Thus the more mature individuals, the more deaths there will be, in turn causing less mature individuals. The loop cannot escalate, we call this balancing. The escalation of the breeding cycle is kept in check by the mortality cycle, and the whole system will balance in the end. Increase in the cause that lead to increase in the result is identified with a +. In a loop, if the result of an increase at the source is returned as an increase at the source, then this is called a reinforcing loop (R). Such a loop will escalate if it is not heeded by other factors. If the result of an increase at the source is returned to the source as a decrease, then this is called a balancing loop (B). No natural systems escalate forever, because there will be balancing loops coupled to them. Depending on the issue we may want to promote a reinforcing cycle, by weakening the balancing loops attached to it, alternatively we may want to limit it by strengthening the balancing loops or attaching new ones. The SUFOR project has been characterized by good interdisciplinary work between the Swedish School of Forestry at the Agricultural University, the Ecology departments at the faculty of Natural Sciences and and the Chemical Engineering department at Lund Institute of Technology. This may seem trivial, but even if sufficient scientific cooperation is usual at most universities, truly integrated research work is indeed not, anywhere. In SUFOR this was achieved, much by building personal networks and strongly promoting informal working teams. Much of what you find in this report has not been invented here. Indeed, that was never the intention to base the whole programme effort all on original innovations, the present report is intended as a synthesis based to a large degree on

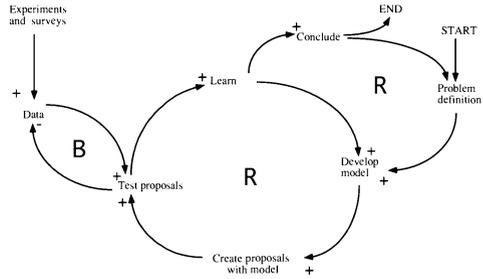


Figure 1.3: *The learning loop is a fundamental concept in the SUFOR Programme. The work is problem oriented, the work started by defining the problem and by formulation present knowledge as a mental model, which in its turn defined the data to be collected. Only after that has been documented, is data gathered, and very selectively with respect to the problem and what the defined model requires to test the hypothesis.*

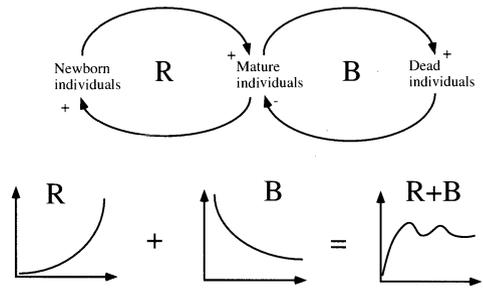


Figure 1.4: *Example of a causal loop diagram for a population. Increase in the cause that lead to increase in the result is identified with a +. In a loop, if the result of an increase at the source is returned as an increase at the source, then this is called a reinforcing loop (R). If the result of an increase at the source is returned to the source as a decrease, then this is called a balancing loop (B). Delays are indicated with //. Below we have drawn the expected reference behaviour of each cycle and then added them to a system behaviour pattern.*

knowledge generated elsewhere, merged with innovations and knowledge developed within the SUFOR programme. Thus, at every turn and question, old knowledge from earlier experiences and data from the literature will be called up, blended with our ideas, inventions and models and pushed forward into new solutions and answers. The report contains several literature reviews and many new syntheses attempting to adopt a wider perspective than earlier. This merging of knowledge from many sources is intentional and desirable. We have stated our conclusions based on our analysis of the available information and the experiences we have made. Everything in the report is not always politically correct, the reflection of our findings were judged to be more important. This implies that we sometimes make statements that may appear to be different than the usual practice.

Harald Sverdrup and Ingrid Stjernquist
in Lund on April 1st, Anno Dei Gratie 2002

1.2 Foreword by the programme director

There is a strong national and international society demand for the multiple use of forests, and retaining or improving biodiversity has been defined as an additional goal along with a high and sustainable productivity of forests. These demands immediately raise a number of practical and scientific questions of how this can be implemented. Are current forestry practices sustainable? In what ways should they be improved? There are many indications today that forest productivity has reached levels that cannot be sustained long-term. A number of environmental factors are gradually changing the conditions for practicing forestry, many of these being factors of human origin. Will it be possible under these circumstances to introduce new forestry practices, including those aimed at sustainability? What are the short term costs of this for landowners and for Sweden, and when will the more profitable future arrive? Can we afford to not begin thinking about sustainability? All these questions and the many different and complex long-term interactions of processes in forest ecosystems and in society that need to be taken into account force scientists to work and think in terms of dynamic system models. It is impossible to combine productivity, biodiversity, nutritional conditions and forest health aspects, as well as economic and sociological demands, without making certain generalizations that are clearly needed in system models. From a natural science perspective the sustainability concept implies above all that favourable soil and water conditions need to be maintained. In combination with biodiversity considerations this has emphasized the need of changes in forest management, in order to preserve the long-term soil fertility. The SUFOR Programme (Sustainable Forestry in southern Sweden) was created to investigate and develop principles of sustainable forestry in southern Sweden in a very broad context, involving sustainability with respect to economic possibilities, nature and society. Within this MISTRA-funded research programme, the first four years from 1996 to 2000, have resulted in a large body of knowledge, being gained through extensive efforts aimed at synthesis. New information has



Figure 1.5: Professor of Plant Ecology at Lund University, Bengt Nihlgård acted as the SUFOR Programme Director. Photo: H. Sverdrup

to introduce new forestry practices, including those aimed at sustainability? What are the short term costs of this for landowners and for Sweden, and when will the more profitable future arrive? Can we afford to not begin thinking about sustainability? All these questions and the many different and complex long-term interactions of processes in forest ecosystems and in society that need to be taken into account force scientists to work and think in terms of dynamic system models. It is impossible to combine productivity, biodiversity, nutritional conditions and forest health aspects, as well as economic and sociological demands, without making certain generalizations that are clearly needed in system models. From a natural science perspective the sustainability concept implies above all that favourable soil and water conditions need to be maintained. In combination with biodiversity considerations this has emphasized the need of changes in forest management, in order to preserve the long-term soil fertility. The SUFOR Programme (Sustainable Forestry in southern Sweden) was created to investigate and develop principles of sustainable forestry in southern Sweden in a very broad context, involving sustainability with respect to economic possibilities, nature and society. Within this MISTRA-funded research programme, the first four years from 1996 to 2000, have resulted in a large body of knowledge, being gained through extensive efforts aimed at synthesis. New information has

been combined with old, and by incorporation of new data from field research and from system insights that have been gained, new computer models are developed. We have conducted or commenced field experiments, made use of the results of many new and earlier laboratory experiments, and analysed thoroughly the practical and stochastic problems to be solved, the management problems involved and the new and promising possibilities. In the SUFOR Programme we have been able to establish quantitative sustainability limits based on the use of multiple criteria for the multiple production goals set within the natural, social and economic spheres, at scales ranging from plots of a few hectares to large regions hundreds of thousands of square kilometres in size. This is the first time this has been done anywhere and it is an important milestone the SUFOR Programme has reached. An example from this programme is presented. It can be remarked that the sustainable yield that is shown significantly differs from that derived from the site index, or the "bonitet". This emphasizes the need for obtaining assessments of this type on an operational scale.

The present book was assembled primarily for communicating the results and insights obtained within the framework of the SUFOR Programme. Most of the basic principles that were adopted in the programme are described in the book. We discuss the possible solutions that have been identified, and present those that formed the basis of the models that were adopted. The models taken up represent the results of extensive collective efforts, and provide a picture of the collective understanding of the programme team. Thus, for anything found to be erroneous or wanting in our models, one should blame not the models as such, but the incompleteness of our knowledge, our treatment of that knowledge, how we have used this knowledge or the way we have communicated it. No understanding or model of anything can be said to exist in society prior to its being adequately communicated. From a societal viewpoint that which cannot be communicated does not exist ! We wish to emphasize both the strengths and the weaknesses of the syntheses described here, and certainly do not claim to have found a solution to everything. We present what we feel we have learned with the view of its representing "insight or the solutions that accord with best knowledge we have available at present". We reserve for ourselves the right to return at some time in the future to revise any of our proposals, to the extent we consider it appropriate due to our knowledge having increased or become more adequate. Knowledge will form the basis for the future validation and implementation phases that we aim at. Phase II of the SUFOR Program will continue throughout the period 2001-2004, being based on the results of the first period.

Bengt Nihlgård
Lund, April 1st, 2002

1.3 SUFOR Programme formulation

1.3.1 Problem description

The need of forest system being sustainable as a system for producing biomass, for preserving biodiversity and for maintaining a semi-natural system for human recreation and inspiration has been brought to public attention. The arrival of the Brundtland Report, the adoption by the Ministerial conference at Rio of Agenda 21, and many years of international research on acid rain have drawn attention to the problem of the sustainability of forest growth within forest management, and to interactions between manipulations of the forest system with pollution stress. However, the problem of sustainable forest use is a large and complex issue, encompassing all aspects of mankind and of society's use of forests. In Sweden sustainable forest management was not necessary as long as harvests were very low as compared with the maximum sustainable production capacity, and a large part of the forest area was only marginally affected by human activities. Historically, modern Swedish forest management

started slightly more than 100 years ago. From the start until very recently it was focused on increasing the productivity of forests, and was highly successful in this respect. Forest management was very much the result of external influences and the introduction of forestry schools in central Europe some 150 years ago. Forest production at present is essential to Sweden as a source of raw materials and of income. This has resulted in the number of trees per unit land area being increased, and in the increasingly effective management of stands and the promotion of tree age distributions that suit the harvesting needs that are planned.

Modern forestry developed hand in hand with a gradually increasing problem of pollution, however, since the start of the industrialization in the 1840s, human activities have resulted in steadily increasing emissions of sulphuric oxides, nitric oxides and ammonia, increasing the atmospheric deposition of nitrogen over time. A 5-fold increase in the nitrogen available for growth occurred at the same time that improved forest management methods increased the amounts of stemwood harvested in the forests two- to three-fold. The results of the Swedish Forest Inventory show that total forest growth and the standing stock of timber to have increased steadily since the first assessments of them were made in 1920. Since the increase in growth appears to be stable and uninterrupted up to the present day, it can appear as if there is nothing to worry about. In 1995, the growth of stemwood in Swedish forests was approximately 95 million m^3 year⁻¹, the annual harvest being approximately 70 million m^3 stemwood (SOU 76:1992). Prehistoric forest growth must have been much slower than that at present, the harvest also being much smaller in earlier times. One can compare historical times in Sweden with present conditions in the Province of British Columbia in Western Canada, where nitrogen deposition ranges from 0.5 to 3 kg N per hectare a year, and the forest is slow-growing and strongly nitrogen limited. Before 1800, prior to industrialization, the situation there must have been comparable to that in Sweden. The harvest of timber in Sweden increased from an estimated 20 million m^3 year⁻¹ in 1840, at the beginning of industrialization, to a recorded 40 million m^3 yr⁻¹ in 1920. Much of the increase in harvest was possible by greatly improved forest management, soil drainage in large areas and afforestation of what had been open land earlier. We would estimate that the nitrogen deposition in 1800 was approximately 2 kg ha⁻¹yr⁻¹. In 1900, nitrogen deposition started to increase significantly, to peak at 25 kg N ha⁻¹yr⁻¹ in southern Sweden and at 12 kg N ha⁻¹yr⁻¹ in the north. It is expected to decrease slowly in the future as an effect of the last LRTAP protocol. From 1850 to 1930, forest management improved tremendously. During this period it was learned how trees are to be managed, and forestry handbooks were made. Much of the pioneering work in this area was started in Prussia in Germany, but Sweden quickly followed suite. The extensive old growth was mined

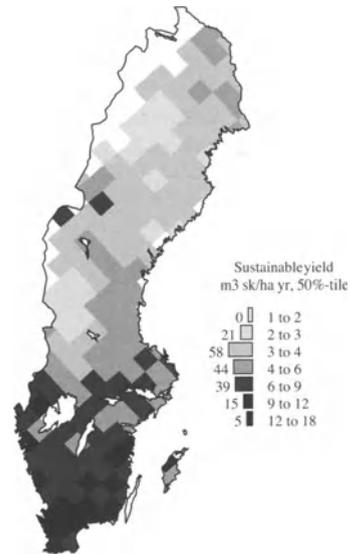


Figure 1.6: In the SUFOR project, performing approximate calculations of sustainability was one of the goals. This goal has been achieved, having succeeded in determining quantitative sustainability limits on the basis of multiple criteria for production goals. These were set for the natural, social and economic spheres, on scales ranging from plots of a few hectares to large regions of hundreds of thousands square kilometres. This is an important milestone reached by the SUFOR Programme. The maps show the maximum amount of stemwood that can be harvested if simultaneously the nutrient budgets for magnesium, calcium and potassium will be kept in balance.

from the forest, and replaced with fast growing uniform stands, mainly Norway spruce of Romanian and White Russian provenances. The old growth that was harvested was taken from mature forest areas, that were declining in total rate of forest growth. Effective management saw to it that the rotation time for replacement of a forest stand became shorter, 60-80 years, and the stands denser, maintaining the forest stands in their most productive phase. As soon as a stand matured and declined in growth, it was harvested. Thus, the age structure of the forest as a whole in Sweden was changed, virtually none of the old, mature and slowly growing old trees that were found in natural, non-harvested forests being left. One should realize that Swedish forests represent a fully developed cultured landscape, in which the tree species, the tree density per unit area, the growth rate and the age structure are almost fully controlled and designed by man. Foresters have been very clever in making forests appear to be natural, at least most of the time, even though they are not. Today, only 4% of the total productive forested area of 267,000 km² in Sweden is covered by truly natural and untouched forests, or "Urskog". In Southern Sweden every other forested area has been logged over at least once, and often twice and in many places is now in its third period of managed rotation. There are another approximately 50,000 km² of wilderness and natural protection areas in Sweden, however, that are closer to natural undeveloped forest areas, even if they are today classified as "unproductive". From the standpoint of natural conservation of resources, these lands are of considerable interest.

Forests are subjected to a considerable number of stress factors. Most of these are natural, others are related to forest management or to industrial pollutants. Various industrial pollution stresses are particularly worthy to note in Southern Sweden, e.g. the effects of ground level ozone, a high nitrogen oversupply, acidified soils caused by acid rain, and effects of global climatic change. Other stress factors, probably of less importance, are the load of the heavy metals lead, mercury and cadmium. Climatic changes are suspected to cause temperature variations and abnormal water conditions, changed conditions for the winter hibernation of trees, putting additional stress on the ecosystems involved. Several of the stress factors may interact and under certain conditions the final effect can be amplified. Modern forest management methods sometimes causes considerable stress on the ecosystem; root rot and several types of insect problems can be considered to be the result of certain types of management practices, and would not likely be expected as problems in natural forests. Changes in the basic conditions of physical and chemical climate, and in nutrient conditions, create new environments for pests and pathogens, leading to "second generation new forest damage". In SUFOR, models are being developed to account for these as well as for interactions when several stressors act simultaneously. Many additional stresses are due to man's activities in the forest, such as recreation, harvesting, hunting, transportation, mechanical disturbances, even soil plowing. All Swedish forests, including those considered natural, are under some degree of anthropogenic influence with respect to species composition and basic conditions for growth. Some 85% of the forest area is used in some way for commercial biomass production, thus in reality being a cultured landscape.

- It is possible to over-exploit the nutrient resources of any forest soil by unwise use of very efficient forestry management methods. The current growth rate of forests are not always correlated very closely with their long-term resource-supply capacity (nutrients, soil structure, and hydrology), implying growth to be sub-optimally allocated with respect to maximum sustainable yield, and also that the growth that is presently achieved cannot necessarily be harvested in a sustainable way.
- Forest systems may be utilised in an unsustainable manner for long periods of time before symptoms of unsustainability become obvious. Symptoms of unsustainability overlap to a high degree with symptoms of pollutants and with symptoms of natural variations in basic conditions. Since several properties of biodiversity change only slowly (over decades or even centuries), irreversible damage may be done long before its effects can be observed. This makes predictive capability important

- The yield tables and prediction methods currently employed were developed under conditions differing from those found today (the climate was colder, less nitrogen was deposited, soil pH and base saturation were higher, and the forests were less well managed). An increase in environmental pollution, chemical changes in the soil and the potential for changes in temperature and in the availability of water, increase the uncertainty in estimates of growth and of yields.
- Swedish forest units are dominated to varying degree by a mono-culture in terms of tree species and by monotonous management. This results in system's having low degree both of species biodiversity and of genetic diversity within separate species. The low variability results in a narrow range of resistance to different types of stress and of pests, increasing the risk for substantial effects of factors considered relatively harmless in more diverse systems. Mono-cultures are especially sensitive to changes in climate.
- The monotonous forest culture promoted by the current forest management paradigm, has led to an impoverishment both of vegetation cover and of the biodiversity, often resulting in monocultural genetic deserts. The devastation of the admixture of birch in conifer forests and the obliteration of large tracts of broadleaf forest areas, which later have been substituted by monocultures of Norway spruce, has changed the character of the landscape and substantially lowered the nutrient sustainability and the resistance to atmospheric acid pollution. The gradual change towards plantations and tree farming causes higher and higher losses of biodiversity in the stands, enhancing the importance of biodiversity carried by other landscape elements.
- Neither monotonous forest management, nor the presence of monocultures in large areas are popular with the general public, the monocultures involved being regarded more as plantations than as forests. This will all come in conflict with the prevailing natural conservation paradigm as known by the public. The elimination of different types of deciduous trees from large areas tends to be regarded by the public as aesthetically unacceptable and as being an expression of a lack of environmental concern.
- The planning of forests in terms of achieving business profits is partly disconnected at present from environmental constraints and from considerations of biodiversity in the implementation of forest management tools. Appropriate tools for the management of forests in the terms of profits, infrastructure, sustainability and environmental effects are lacking. Specifically, initial forest regeneration costs have traditionally been allocated to initial investments in the establishment of forests, rather than being seen as a prerequisite for successful harvest and being written off directly against harvest profits. In small scale forestry this is actually happening.
- Present economic analysis of Swedish forestry is dominated by the large industrial corporations. Although their philosophy is well suited for forestry in northern Sweden, it does not reflect the truth of matters in southern Sweden where, 85% of the land and 50% of the new growth are held by small private owners in lots of 100 to 1000 hectares. For these the presently applied economic rules and assumptions only apply partly or not at all. If realistic predictions are to be made, new approaches and new models must be developed, involving longer time-perspectives adapted to the accounting tradition of smaller private farms engaged in a variety of economic activities.
- Nitrogen accumulates in the forest floor at a rate of 3-12 kg/ha yr in Swedish forests. Although at present, this does not appear to cause large observable direct effects, it is likely that this is one of the causes behind ground vegetation changes. If the changes are adverse or not, is not yet determined. In the long run, the accumulation cannot go on forever, and at some point the excess nitrogen may start leaching.

- Swedish forests are not in balance with respect to carbon. Some methods of management are suspected of increasing the decomposition of organic matter, and release large amounts of CO₂ and dissolved nitrogen from the forests. Other forest management methods lead to large amounts of carbon and nitrogen accumulating in the forest floor. It is not known just what causes a forest system to go from accumulation to depletion, this matter not yet having been subjected to qualified systems analysis. In order to get the forest to actively sequester net amounts of CO₂ from the atmosphere, decomposition needs to be artificially brought about by means of appropriate forest management design.
- In southern Swedish forests many of the basic conditions assumed to be constant in the forest ecosystem, may start to change considerably if the climate should change in ways predicted by present models.
- There are several major problems in regard to forest health, forest growth and environmental pollution in southern Sweden. Some of these problems are well known and have been documented in detail by the Swedish Forestry Board and associated organizations, whereas others, though possibly equally significant, have for various reasons not yet been recorded or identified as important problems.

It is no longer regarded as self-evident that there always will be a sufficient supply of essential nutrients when very large volumes of biomass are removed from the forest system each year. To understand the nature of the problem, one need to analyze what the consequence would be in the long run if larger amounts of nutrients are consumed in forests than can be supplied to them. This concerns both bulk nutrients such as nitrogen, phosphorus and base cations, as well as trace elements under certain circumstances. The total nitrogen deposition in Sweden varies from 25 kg N ha⁻¹yr⁻¹ in the south to less than 5 kg N ha⁻¹yr⁻¹ in the far north. The background nitrogen deposition historically available was probably in the range 1.5-2 kg N ha⁻¹yr⁻¹. Ongoing work on limiting pollution may make it possible to significantly reduce nitrogen deposition in the future, recreating a situation with strong nitrogen deficiency. All plants require relatively fixed ratios of nitrogen to individual base cations in their total nutrient uptake flux in order to be able to produce needles, bark, branches, stems and roots, the uptake elasticity in building plant parts not being particularly large. If base saturation was reduced to a very low value, the soil solution in most locations would obtain high concentrations of Al that potentially cause problems for tree roots, and low stability of the ecosystem. The base saturation is important as a short-term resource to cover high needs of base cations. Such "loans" can be repaid in other seasons

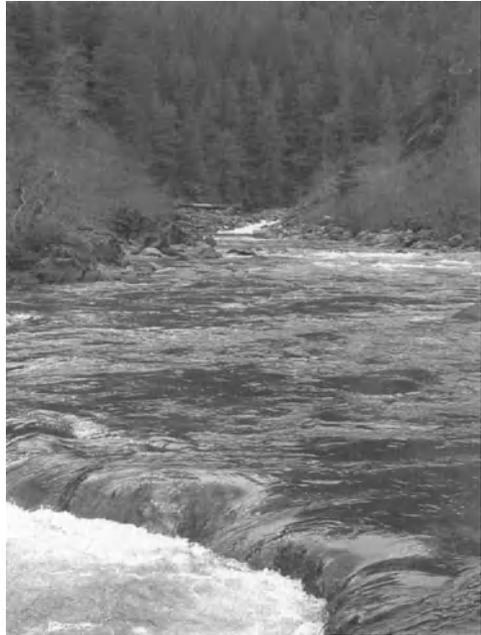


Figure 1.7: *The forest, the water and the catchment are all parts of the ecosystem, and they all affect each other. Thus management may affect the water quality of the runoff or the soil status of the catchment.*

when the uptake need is lower than the supply from weathering and deposition. In the past, several of the industrial forest management practices lead to conflicts with the public, and with what the society perceives as permissible forestry practices. A good example is that of the large clearcuts created in Sweden, these being viewed as negatively from an aesthetically standpoint, and worsening public recreational facilities, as well as sometimes creating soil erosion. These have called for public outcries and talks about "mismanagement". This is a clear example of a management method that may well have been justified from the standpoint of maximizing production, but was socially unacceptable. Foresters in large forest companies did not listen to the rising public outcry, but elected politicians changed the forest law, and this management ended. This also illustrates some of the limits to property rights, and the fact that the public is a major stakeholder of various immaterial assets in any large forest area. One can also notice that the degree of conflict with public wishes, and the degree of insensitivity to public opinion has tended to be roughly proportional to the economic size of the forest owner involved. In the long run large companies may have a less suitable management structure for adopting to total sustainability views than the small or medium sized economic unit. Large forest owners seems historically to have had far more difficulties in managing their properties more sustainable than smaller owners and small family farms (Marchak 1997; Drengson and Taylor 1997).

1.3.2 Hypothesizes of the programme

We are planning for a future in which forestry, thinking in terms of 100-year or more cycles, has to be prepared for altered climatic conditions, very likely involving an increasing average temperature, which favours deciduous trees. Simultaneously during the coming decades the nitrogen deposition will continue to be high or it will slowly decline, and the levels of ozone will be excessive. We foresee an increasing global market for high quality hardwood, as well as for softwood and fuel wood. We believe that silvicultural methods practiced in southern Sweden in the future will involve greater use of self-regenerating forests and modified final-felling, and will favour trees possessing properties of highest quality. Deciduous trees will play an increasing role in forests as stabilizing elements and as components for enhancing natural nutrient supply use efficiency in production. Nature conservation measures will often be taken at the stand level and be planned on a landscape scale, use being made of GIS-maps. Biodiversity will be much better maintained when such methods are employed. Problems of soil acidification and nutritional imbalances will be avoided by returning wood ash and/or minerals to the forest stands in question, thus stabilizing biogeochemical cycles. Forest owners will learn how to act when stress symptoms appear in trees, including symptoms that reduce stem quality. Another, contrasting development in forestry will be silvicultural methods that favour forests for intensive production of pulp and bioenergy in certain areas. Sewage and waste products from society will be used increasingly as fertilizers. The recycling of nutrients in this way can serve to balance the continuous losses caused by yield and leaching. Society will tend to accept certain 'cultural' forests that have high yields, as well as semi-natural flora and fauna, in light of the more environmentally adapted forestry methods employed in most other areas. The new methods developed will create more varying landscape, with high degree of variation in tree species, flora and fauna, matters which society will very much appreciate. Improved soil qualities and more stable wood production together with increased resilience of trees to climatic change and other forms of stress are all matters that will be positively received by the many forest owners found in southern Sweden. Wild game will be even more important than it is today, compensating for the possible negative effects caused by an increased browsing and increased tourism. The following hypotheses are common to the programme as a whole:

1. The increase in the proportion of deciduous trees in conifer forests, and the avoiding of large clear-fellings and plantations, will increase sustainability through greater soil fertility and biodiversity being achieved. Mixed stands will exploit the increase in soil penetration by a multi-species plant community, providing better joint access to in-soil nutrient pro-

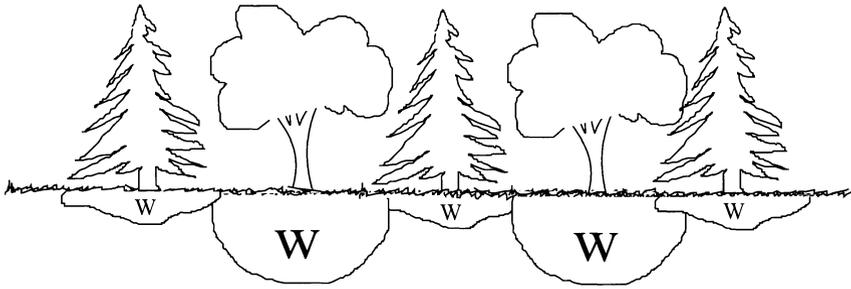


Figure 1.8: *In Sweden it has been noted that Norway spruce has shallow fine roots, that Scots pine and birch have somewhat deeper roots than Norway spruce, and that beech and oak tend to be deeply rooted. In the Jämjö case study, this could also be shown to occur in the field. This forms the basis for one of the hypotheses of the programme, that a tree with deeper roots has access to more weathering products than a tree with shallow roots, provided that the depth to bedrock or the depth to groundwater allows this difference to develop.*

duction. Deciduous trees generally have deeper root systems than Norway spruce, giving them access to greater amounts of weathering products. Roots of trees used commercially in Sweden seldom cross into the groundwater, but tend to stay above the groundwater table by a small margin (Grönare Skog, pages 66-67 Skogsstyrelsens Förlag 1999, Grundbok för skogbrukare, pages 116-119, Skogsstyrelsen (Swedish Forest Directorate) 1985). See further Fig. 1.8.

2. Natural regeneration will be more sustainable in an overall system involving lesser use of resources and lesser disturbance of the soil than is the case of artificial planting. Natural regeneration requires less soil ploughing and disturbances from heavy equipment, and involves more in situ genetic selection at a particular site. In the scenarios investigated, regeneration will be found on a continuous scale ranging from full natural regeneration to full planting. The greatest advantage of natural regeneration can be regarded as being economic, by eliminating the cost of planting.
3. The potential for forest health to improve will be greater when nutritional balance is achieved, allowing more resources to be allocated to the defence system.
4. The susceptibility to acid pollution and to nitrogen deposition will be reduced through forest management's adapting to multi-species forestry, and to forestry as a whole being more profitable.
5. Soil fertility will be retained by the addition of missing macro- and micro elements, in accordance with long-established farming traditions, unless harvest is adjusted to natural supply rates. This will ensure a sustainable productivity and better maintain biodiversity and forest health.
6. The changes in large scale transboundary pollution and the global climate changes that are anticipated, will alter the functioning of different tree species in forest in such a way that a rethinking of present forest management methods will be necessary on order to preserve profitability, while keeping at the same time within the constraints of sustainability.
7. Forest management will be adapted in such a way as to have beneficial effects on surface water.

8. Minimum biodiversity and a sustainable potential for biodiversity regeneration will be ensured by a sufficient number of undisturbed forest areas being maintained of a size and area well above what can be considered a minimum or critical level.
9. Modelling natural variations in forest productivity, forest-damage effects, biogeochemical cycles and biodiversity will contribute considerably to an understanding of how to manage forests in a sustainable way.
10. Simple operational methods will be developed for assessing nutrient (P, Ca, Mg, K) sustainability on private forest properties (50-2,000 hectares), methods very low in costs and useable for periods of 200 years or more.
11. Since the conditions for sustainability will vary considerably over time and will vary from place to place, marked changes in physical climate, in pollution condition and soil status will occur the next 250 years. In line with this expectations the programme needs to develop flexible methods and models for determining the degree of sustainability in light of the natural, social and economic factors that can affect it.
12. For a small family farm, mixed forestry for the production of both pulpwood and high quality hardwood timber will tend to be more profitability and more sustainable long-term, than forestry involving a Norway spruce monoculture.
13. Biodiversity can be better preserved and protected in a mixed forest management regime than in a monocultural conifer regime.

The overall hypotheses have been broken down to more detailed hypotheses which have been set out in the programme plan. Some of these hypotheses are tested in this report.

Chapter 2

On Swedish forests

Per Stjernquist, Ingrid Stjernquist and Harald Sverdrup

2.1 The forests in southern Sweden

2.1.1 Forest characteristics

The forests of Sweden cover 22,613,103 ha, which is more than half of the land area of the country, see Table 2.1. As the climate changes from south to north, the biotic zonation covers the nemoral, boreo-nemoral and boreal zones (Sjörs 1965). The border between the nemoral and the boreo-nemoral zones is defined by the southern distribution limit of Norway spruce. Most of the southern counties of Scania, Blekinge and Halland are within the nemoral zone, Fig. 2.1. In this region, Scots pine and deciduous tree species are natural forest elements, but forest management and the better economic outcome for conifers during the last century have created a high percentage of Norway spruce, Table 2.1, 2.2. Beech grows mainly in the counties of Scania, Blekinge and Halland, but scattered stands of beech are frequent north of this area (Lindgren 1970). The boreo-nemoral zone reaches north to the middle part of Sweden, its northern border being defined by the forest limit of *Quercus robur*. This zone comprises the regions of Svealand and most of Götaland, Fig. 2.1. The southern Swedish forests are defined as the forest area of Götaland and Svealand.



Figure 2.1: Location map over southern Sweden showing the SUFOR experimental sites and the positions of the various provinces. Experimental sites: 1. Gårdsjön, 2. Skogaby, 3. Tönnersjö forest research park, 4. The Söderåsen ridge, 5. Maglehem, 6. Fulltofta, 7. Jämjö, 8. Asa forest research park, 9. Stenbrohult, 10. Hornsö, 11. Kvill National Park. Counties: A. Scania, B. Blekinge, C. Kronoberg, D. Halland, E. Jönköping, F. Kalmar, G. Västra Götaland, H. Östergötland.

| | Forest area | Scots pine | Norway spruce | mixed conifers | mixed ¹ species | deciduous ² trees | Valuable ³ hardwood |
|----------|-------------|------------|---------------|----------------|----------------------------|------------------------------|--------------------------------|
| Scania | 34 | 11.4 | 42.4 | 3.5 | 5.0 | 17.1 | 17.2 |
| Blekinge | 70 | 6.5 | 53.3 | 7.8 | 7.7 | 7.8 | 14.2 |
| Halland | 55 | 17.6 | 48.8 | 10.2 | 7.1 | 8.6 | 5.5 |
| Götaland | 57 | 25.2 | 39.1 | 14.1 | 6.9 | 7.9 | 3.6 |
| Sweden | 55 | 40.1 | 27.4 | 15.7 | 7.8 | 5.1 | 0.8 |

Table 2.1: *Forest land areas of different stand types in percentage in southern regions of Sweden in 1994-1998 (Skogsakta 2000).* ¹: The stand must consist of 40-60% deciduous trees. ²: The stand must consist of at least 70% deciduous trees and less than 50% valuable hardwood. ³: The stand must consist of at least 70% deciduous trees and 50% valuable hardwoods. The valuable hardwood species are *Quercus spp.*, *Fagus sylvatica*, *Ulmus spp.*, *Fraxinus excelsior*, *Tilia cordata*, *Acer platanoides*, *Carpinus betulus* and *Prunus avium*

In this area, the landscape is characterized by a mosaic of forest stands, agricultural land, mires and lakes. The forests are managed primarily by private forest owners who each have 20 to 400 ha of forest land, this creating considerable variation in stand characteristics. Historically, deciduous stands and valuable hardwood species have been more frequent due to the large amount of utilities from the forests which contributed to the economy of the farms. The valuable hardwood species, *Quercus spp.*, *Fagus sylvatica*, *Ulmus spp.*, *Fraxinus excelsior*, *Tilia cordata*, *Acer platanoides*, *Carpinus betulus* and *Prunus avium*, were concentrated around the hamlets, often pollard or coppiced, and used for fodder and constructions. Beech and oak have, in many areas been important for products sold outside the farm such as pigs, potash etc.

The total timber volume in Sweden 1994-98 was 2,829 mill m³sk. Of this, 55% grew in the southern part (Götaland and Svealand) on 10,235 mill ha of forest land, Table 2.2. Since 1920, the Swedish timber volume has increased by about 50% due mainly to an increase in conifer plantations (Skogsstatistisk Årsbok 2000). The standing volume of spruce is now greater in the southern part of the country than for Sweden as a whole.

The forest management methods employed, such as clear-felling and planting, have contributed to the development of monocultures, Table 2.3. Mixed stands have been considered difficult to manage. Another important factor, in recent years, for the increase in pure conifer stands has also been the strong impact of grazing by deer and elk on the survival of deciduous tree seedlings. The need for fencing has made the costs of replanting deciduous species high compared to those of spruce. The valuable hardwood species often grow in mixed stands, only beech, oak and ash form relatively pure stands due to the historical utility of these species. In the 1970s continuous stands of beech and oak covered only 100,000 ha and continuous stands in which valuable hardwoods dominated 10,000 ha. Continuous stands are here defined as forests in which there is a hardwood stem basal area of more than 70%.

Valuable hardwoods mixed with other species, e.g. birch, aspen and spruce, cover an additional 200,000 ha (Almgren et al. 1984). A special Forest Act for the management of valuable hardwoods was found to be needed to preserve these forest types. The 1984 Valuable Hardwood Forestry Act provided a definition of valuable hardwood forest types as well as directions on forest management, including clear-cutting and replanting. The Act was based on scientific, cultural, social and silvicultural considerations (Almgren et al. 1984, SOU 1992:76). Since 1992, this Act has been included in the general Forestry Act of Sweden. The characteristics of

| | Scots pine | Norway spruce | birch | alder | oak | beech | valuable hardwood |
|----------|---------------|------------------|-------|-------|-----|-------|----------------------|
| Scania | 11.5 | 47.9 | 9.4 | 5.5 | 5.6 | 13.4 | 3.0 |
| Blekinge | 9.8 | 61.4 | 7.2 | 1.4 | 6.8 | 7.4 | 1.7 |
| Halland | 21.3 | 58.9 | 8.5 | 1.5 | 3.9 | 3.1 | 0.4 |
| Götaland | 29.9 | 51.4 | 8.6 | 1.8 | 2.8 | 1.8 | 0.9 |
| Svealand | 43.0 | 43.1 | 8.1 | 1.4 | 0.3 | 0 | 0.1 |
| Sweden | 38.2 | 44.4 | 10.3 | 1.1 | 0.9 | 0.5 | 0.1 |

Table 2.2: *Timber volumes for different tree species in percentage in southern regions of Sweden (Skogsstatistisk Årsbok 2000).*

| | clear-cut ¹ | planting ² | soil scarification ² |
|----------|------------------------|-----------------------|---------------------------------|
| Scania | 2,100 | 2,300 | 2,300 |
| Blekinge | 1,900 | 2,000 | 700 |
| Halland | 1,700 | 1,800 | 700 |
| Götaland | 42,000 | 53,000 | 30,100 |
| Svealand | 52,500 | 28,800 | 39,100 |
| Sweden | 206,000 | 132,500 | 160,200 |

Table 2.3: *Forest management in southern regions of Sweden, ha. 1; in 1999, determined for clear-cut areas larger than 0.5 ha. 2; a 3-year average during 1996-98. (Skogsstatistisk Årsbok 2000).*

the forests in southern Sweden have been molded by continuous use of farmers during the last 1,000 years. The forests are more of a cultural heritage than a wilderness. With the exception of some areas with large estates, most of this region has traditionally been dominated by family farms, often freeholders or Crown farms. At the beginning of the 18th century, 68% of the Swedish land outside the mountainous area in the north was owned by freeholders or Crown tenants and at the end of the 19th century, 60% of it was owned privately by farmers. The partitioning of common land according to a legal act in 1827 scattered the villages and forced 34% of the farmers to reconstruct their farms on what had formerly been common land outside the old villages (Gadd 2000). The resulting small hamlets and single farms dispersed over the landscape, have transformed southern Sweden into a mosaic of forests, meadows and arable land that still is evident. The economic and technical changes during the 20th and late 19th centuries, however, caused a strong impact on the southern Swedish landscape. Old meadows and grazing land were replanted with forests, often conifers, and small farms were abandoned or were combined into larger units. Today, the areas close to the existing hamlets, in which there is a mixture of arable land, meadows and small woodlands are very important for maintaining a high biodiversity. To preserve the existing biodiversity of Swedish forests, special guidelines were included in the 1992 Forestry Act. Besides that, about 6% of the productive forest land has different degrees of protection, including national parks and nature reserve areas, Table 2.4. However, in the southern region, only 1.5% of the productive forest land have any kind of

| | Total protected ¹ | Protected ² |
|----------|------------------------------|------------------------|
| Scania | 0.7 | 2.5 |
| Blekinge | 0.6 | 2.1 |
| Halland | 0.5 | 1.4 |
| Götaland | 0.6 | 1.5 |
| Svealand | 1.8 | 4.0 |
| Sweden | 3.6 | 6.0 |

Table 2.4: *Areas of protected productive forestland in southern regions of Sweden, in percent. 1: 1996. 2: 1997 (Skogsstatistisk Årsbok 2000).*

protection. Another method to protect areas with very high biodiversity is the creation of key-habitats, which cover around 1% of the forest land (Skogsstatistisk Årsbok 2000). The different systems of forest certification also include guidelines to stimulate an environmental sound forest management. In 1999, 10 mill ha of forest land had some kind of certification.

Owing to the great diversity in forest management, both historically and today, southern Swedish forests represents a unique experiment for studying long-term sustainable forest management. The privately owned forest land has often belonged to the same family for generations and the knowledge of the soil characteristics, the changes of tree species and the management methods used during the last century are extremely valuable both for evaluating scientific results and for modelling a sustainable forest management in the future.

2.1.2 SUFOR experimental sites

The projects within the SUFOR programme have used many experimental sites in southern Sweden, both on privately owned land and in forest research parks, to study differences in forest management, soil characteristics and climate. To simplify for the reader to follow the discussions in the following chapters, a location map of the counties and experimental sites involved is presented in Fig. 2.1.

2.2 Forest ownership in Sweden

2.2.1 Ownership structure

The dominant forest owner categories in Sweden are private forest owners and the timber companies, Table 2.5. Today, there are about 250,000 private forest holdings in Sweden altogether. They are unevenly distributed over the country, however. In the southern part of Sweden as much as 78% of the forestland are privately owned as compared with 41% in the northern part. On the other hand, far more company forests, 50%, are found in the north as compared with only 13% in the south. The State and other owner categories control about 10% of the forests. Of the forest area that is privately owned, 32% consist of farms that include both forest and agricultural land (Statistical Yearbook of Forestry 2000). Up to 1960 nearly all private forest owners were farmers. Historically, most Swedish farmers have owned both forest and agricultural land and have used the resources of both to create economically sustainable businesses. Forests have been looked upon as a resource for multiple use, not only for the requirements of the farm but also for selling refined products on the market. Examples of such products are timber, coal, tar and potash and also cattle, pigs and horses that grazed on the forestland during the warm season. The choice of products to refine depended to a great deal on the soil

| | Private owners | Companies | The State | Forest land |
|----------|----------------|-----------|-----------|-----------------|
| | % | % | % | km ² |
| Norrland | 41 | 50 | 9 | 123,780 |
| Svealand | 50 | 37 | 13 | 53,040 |
| Götaland | 78 | 13 | 9 | 49,310 |
| Sweden | | | | 226,130 |

Table 2.5: *Forest area by ownership categories during 1994-98 given in percentage of forestland in the northern (Norrland), middle (Svealand) and southern (Götaland) parts of Sweden. The State category includes ownership by either the State and the Church (Statistical Yearbook of Forestry 2000).*

and vegetation characteristics which differed between regions. Since many of the farms have belonged to the same family for generations, knowledge of the productivity of the land and of the limits to a sustainable yearly outtake was important.

As a result of traditional ownership, 80% of the private forest holdings in Sweden today are 20-400 ha in size, with a mean of 20 ha (Statistical Yearbook of Forestry 2000). By comparison, 95% of the company forests and 83% of the State forests are above 1,000 ha in size. Until about 1960, Swedish forest policy recognized the fact that in most parts of the country a combination of agriculture and forestry was a necessary condition for the existence of farms. As the proportion of the employed population engaged in farming and forestry decreased from 14% in 1960 to 3.6% in 1990, the policy was changed radically pressure being mounted for producing as much and as cheap wood as possible in the interest of the timber industries. The Forestry Act of 1979 represented the height of this policy. More recent legislation in 1993 restored the balance between the interests of forest owners and those of industries by adding respect for natural values as a legal goal of the same importance as production. The differences in climate, vegetation period and soil conditions within Sweden result in growth period of trees

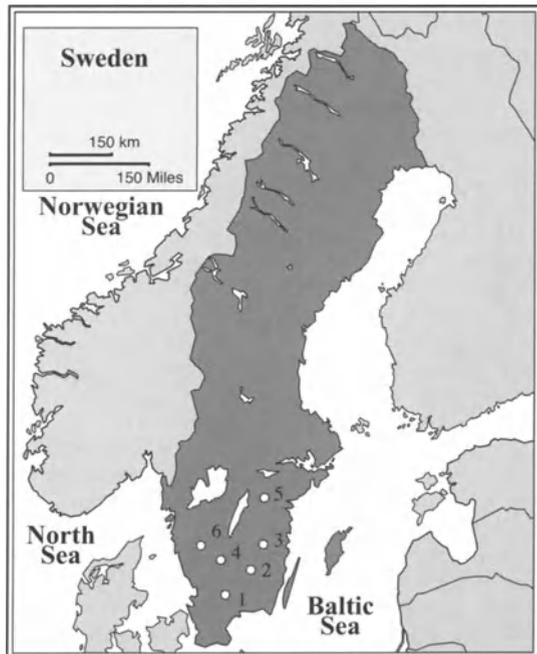


Figure 2.2: *Location of the districts used in the survey of forest owners attitudes to environmental goals in private forestry 1990. All districts were located north of the southern limit for Norway spruce and south of the northern limit for oak (Stjernquist, P. 1992). For numbers see the text.*

| Goals | % |
|--|----|
| Economic yield only | 22 |
| Economic yield, primary goal, regard to natural values secondary | 45 |
| Both goals of equal importance | 24 |
| Regard to natural values the only or primary goal | 9 |

Table 2.6: *Declared goals of private forest owners for their forestry in 1990, (Stjernquist, P. 1992).*

differing between the regions. The site quality class assessment made by the Swedish National Forest Inventory, which evaluates the biomass productivity of the forest stands, give a figure of $3.0 \text{ m}^3\text{sk ha}^{-1}\text{yr}^{-1}$ in the north and $8.7 \text{ m}^3\text{sk ha}^{-1}\text{yr}^{-1}$ in the south, that in Scania being as high as $11 \text{ m}^3\text{sk ha}^{-1}\text{yr}^{-1}$ (The National Atlas of Sweden, The Forest, 1990). However, the measured average yearly growth per ha for the period of 1994-98 is lower, in the north $2.4 \text{ m}^3\text{sk ha}^{-1}\text{yr}^{-1}$ and in the south $6.4 \text{ m}^3\text{sk ha}^{-1}\text{yr}^{-1}$. The timber volume span given for the same period is $85\text{-}172 \text{ m}^3\text{sk ha}^{-1}$. Since private forest owners dominate in southern Sweden, their forestland have a site quality class as well as a timber volume per ha above the average for the region. For example, company forests for Sweden as a whole, have a site quality class of $4.5 \text{ m}^3\text{sk ha}^{-1}\text{yr}^{-1}$ as compared with $5.9 \text{ m}^3\text{sk ha}^{-1}\text{yr}^{-1}$ for privately owned forests. This means that private forest owners is a very important group in work on developing ecologically sustainable forest management (Statistical Yearbook of Forestry 2000).

2.2.2 Private owners and ecological sustainability

Ecological sustainability is a theoretical concept that private forest owners often have difficulties in understanding. During the last ten years, authorities have given general courses in ecological subjects to private forest owners. The contents and results of these courses have not been scientifically examined, however. Certain conclusions regarding the readiness of private forest owners to accept preservation and environmental aims can be drawn, nevertheless, from their attitudes towards forests and their methods of using them. In a project in 1990, including 274 private forest owners from different parts of southern Sweden were requested to write down their goals in forestry and how they treated their forests in practice. The geographical distribution of respondents, who represented holdings of different sizes and a variety of social categories (farmers, residents and absentees), is shown in Fig. 2.2, in which the dots represent cooperative areas of private forest owners (Swedish; skogsbruksområden). The six forest districts investigated in the survey was 1. Sydvirke forest commons (located in northern Scania), 2. Uppvidinge (Kronoberg province), 3. Vimmerby (Kronoberg province), 4. Värnamo (Kronoberg province), 5; Finspång (Östergötland province) and 6. Borås (Västergötland province). No significant differences was found between the different categories. For this reason they are dealt with as a whole. The declared goals in forestry are presented in Table 2.6.

It is possible that Table 2.7 indicates too sharp a distinction between different attitudes. Many of the forest owners who declared economic yield to be their only goal added that they preserved specific areas for reasons such as preservation of "natural forest" or "some wilderness" or "flora and fauna" or simply so as to have something beautiful to look at. There is a clear correspondence between the goals declared and the forestry practices carried out. The different goals of forestry find expression in specific practices. In Table 3 the practices linked with goals regarding natural values appears first and rational practices further down the list. It should be added that the regards to natural values of the forest owners are not exactly the same as

| Practices attached to regards to natural values | % |
|---|----|
| No clear-cutting | 9 |
| Clear-cutting at most 0.5 ha | 8 |
| Clear-cutting at most 1 ha | 33 |
| Growing mixed forest | 94 |
| Saving mature stands | 50 |
| <i>Rational practices</i> | |
| Clear-cutting larger than 1 ha | 50 |
| Using fast growing plants | 44 |
| Prompt felling | 42 |

Table 2.7: *Current practices, 1990, of the respondents, (Stjernquist, P. 1992)*

what modern ecologists try to develop. The forest owners in the investigation appear to attach primary importance to their experience of the forest as a natural and landscape scenery.

As Table 2.7 indicates, the percentages of some ecologically valuable practices are much higher than those of forest owners declaring regard for natural values to be their primary goal or a goal of the same standing as economic goals. One can ask why the affection for the natural state of forests is so widespread among forest owners in Sweden. It is statistically evident that economic and technical explanations alone do not suffice. More irrational explanatory factors needs to be added. It appears from the investigations, and also from other studies, that it is customary in the Swedish rural population to consider farms and forested land as family possessions that are inherited to be passed on from generation to generation. This custom gives rise to a long-term perspective and to a wish to preserve the character of the forest or certain remains of it for future generations. Of importance is also the gradual transition within private forestry from the previous large-scale fellings at long intervals to the small annual fellings of today, often a kind of selection system. This change-over was connected with the proceeding commercialization of the farms that combined agriculture and forestry and which represented the overwhelming part of private forestry. In order to pay the interest and the amortizations of the investments made in land, buildings, and machines, farmers became accustomed to making small annual fellings and to taking care of the forest in such a way that it could also provide a continuous income in the future.

The restricted felling practices together with the existing affection for natural values facilitated the advisory service of the local forest authorities in forestry matters. Previously, the authorities handed down directions, but in the 1930s the forest owners themselves sent in a multitude of applications for service, courses, and demonstrations. A common interest in forestry had been raised. This interest represented a much stronger guarantee for forestry in the future than enforced forest dispositions. How important it is to achieve the acceptance of forest owners should be seen as a key lesson for future policy. The task of transmitting an understanding of ecological sustainability and biodiversity to forest owners rests at the local personnel of the Regional Forestry Boards (Sw. skogsvårdsstyrelser), who have long-term experience in acting as advisors in matters of sustainable forestry. By acting as advisors rather than persons in authority, they have managed to arouse strong interests in forestry methods. In the same way, they will surely be successful in extending the external regard of the forest owners for natural values to understanding of what ecological sustainability requires.



Figure 2.3: *A typical rural landscape in southern Sweden. A pastoral scene showing cows and forest edges at the right and grazing land with coppices, bushes and hedges in the distance. Photograph by Peter Schlyter.*



Figure 2.4: *A roadside in rural Sweden, showing large individual trees, hedgerows and century-old stone fences. Such landscape components are often found between different forest stands and form important components for retaining publically appreciated aspects of biodiversity in the landscape. Preserving such localities at selected spots and as corridors may represent one strategy in the management of biodiversity. Photograph by Peter Schlyter.*

Chapter 3

Defining sustainability

Harald Sverdrup and Mats G. E. Svensson

3.1 Introduction

How to define sustainability has been discussed for a long time. The group assigned to create a "global agenda for change" by the General Assembly of the United Nations in 1983, The Brundtland Commission, initiated the first serious discussions with real political implications. The commission's definition of sustainability has been referred to by diverse bodies as constituting a basis for policies and practices designed to support a society being economically, ecologically, and culturally sustainable: Humanity has the ability to make development sustainable - "to ensure that it meets the needs of the present without compromising the ability of future generations to meet their own needs" (Agenda 21 i Sverige 1997; World Commission on Environment and Development 1987; Holden 1997). This statement is at the heart of sustainability, but leaves open for discussion how the goal as stated can be achieved. Robert Gillman (1981) extended this goal-oriented definition by restating the last part of the sentence above, referring to an old saying; "...do unto future generations as you would have them do unto you." When developing rules and criteria for sustainability, it is important to conceive of them as being basic principles, that insofar as possible are free of value judgments. Sustainability then becomes a property or function which to a large extent should be free of value judgments or cultural biases. The points of departure for defining sustainability are the second law of thermodynamics and the law of mass and energy conservation (Eddington 1987). The aim is to define the basic principles of sustainability in such a way that we may for every situation be able to determine whether a given activity is sustainable or not, since this would require that definitions be precise, unique and functional. It is also important that the rules of sustainability be applied in a realistic way, in order to avoid unfruitful dogmatism and too much observance of political correctness. It should be borne in mind that the demands for natural and economic sustainability must also be sustainable socially if they are to have any practical significance for human society (Gilman 1990; van Pelt et al. 1995; O'Riordan 1988; Dryzek and Schlossberg 1998; Eckersley 1992; Gale and Cordray 1991; Ponting 1993).

3.2 The dimensions of sustainability

Three forms of sustainability need to be defined:

1. Natural sustainability, which defines the maximum long-term use of a natural resource as a source of raw material and energy, the capacity for the use or destruction of waste and exploitation of living organisms.

2. Social sustainability defines the inherent stability of a social organization and its components, the minimum requirements for system resilience to system oscillations, individual rights, limitations and duties for sustainability. It defines necessary gradients and driving forces necessary for society to remain stable, but still respecting individual integrity.
3. Economic sustainability in absolute value terms, derived from mass balance and economic feedback principles.

Certain of the sustainability limitations have precedence before others. In order to be relevant, the parameters of social and economic sustainability for example, need to stay within the boundaries set by the criteria for physical and thermodynamic sustainability of the natural system. There is no exception to the principles of thermodynamics and mass conservation, regardless of the ideology or the reasons one have. Social and economic parameters, in order to remain sustainable, must be kept within the are defined by the criteria of natural sustainability. Thus, a hierarchy needs to be established for the restrictions that the demand for sustainability involve, natural restrictions overruling economic restrictions in the long term, and economic restrictions in the short term, overruling social, even if in the long term the latter may be the opposite. It is important to bear in mind the fact that there are strong feedback mechanisms between these parameters and that they need to be carefully evaluated to determine whether they should be included or ignored and under what circumstances. Economy has an apparent weak feedback from natural limitations at present, particularly since our social systems do not observe and enforce adherence to natural limitations very strongly. Many economic interests at the moment appear to have no bounds set on them whatever by natural limitations. In due time, nevertheless, natural limitations will enforce themselves completely requiring absolute obedience to them, whether we want this or not. A system can exist outside the area of sustainability for a limited period of time. A limited resource is thus often exploited for as long as it lasts, before the natural enforcement of limitations takes place. There are many examples of societies prior the present ones, that vanished because of a lack of understanding of the limiting factors of the availability of natural resource factors in the long run. There are four dimensions in terms of which the sustainability of a system can be considered. These are time, space, matter and structure. The spatial dimension represents how the system is represented in space, what spatial components can be identified, what the spatial borders of the system are and what the links are to other systems across the borders of the system. The temporal dimension concerns the level of resolution for the system dynamics over time, the time step used in analyzes carried out, the temporal events that should be either by viewed singly or be lumped together. The structural dimension, finally, is the level of chosen to describe the processes and the functions in the system, the variables involved and the links between them (Bahn and Flenley 1992; Sabloff 1990). From this one can derive the following basic definition:

The sustainability limit concerns the long-term use of the ecosystem, the intensity of the use being such that the resources available and the structure and functions of the ecosystem are not unacceptably degraded or changed..

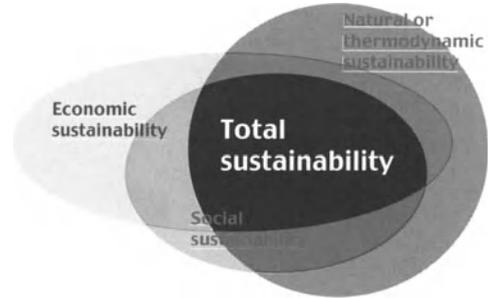


Figure 3.1: *Total sustainability is defined as the area of overlap of all three types of sustainability; natural, economic and social. This common area represents the area of full sustainability. This common area implies greater freedom of choice. Optimizing sustainability means obtaining as a common area as possible.*

A number of corollaries to this definition can be derived. A first corollary that we suggest is the biodiversity criterion:

The biological self-organizing capability of the system must remain intact

This implies that use of the system will be such that after exploitation occurs the ecosystem is able to self-organize its natural function and to self-regenerate. This applies to such matters as the abundance, geographical distribution and genetic variation of a species. The potential for a varied ecosystem needs to be found in terms of the species present must exist and these species must contain the sufficient genetic variation for establishing interacting, but stable populations. The sites should also not have been altered in physical or chemical terms in a way such that an entirely different vegetation than earlier develops. Although it is the natural strategy of a tree to take up all the nitrogen and phosphorus available, when the nitrogen and phosphorus taken up cannot be matched with the amounts of Mg, Ca or K needed to build plant biomass, problems of nutrient regulation may arise because of conflicting signals of simultaneous surplus and deficiency. When the supply of base cations available falls below current needs as set by the current growth rate and the ambient needle mass, less needles can be sustained the following year, which has a corresponding effect on growth. In the long run, there will be no way to avoid *Liebig's law*, based on the universal principle of mass conservation. It can be proven that

The amount of growth that can be harvested is limited by the nutrient with the slowest rate of supply.

The term nutrient is to be understood here in the broadest sense, that is as including such mineral nutrients as N, P, Ca, Mg, K, micronutrients, water CO₂ and energy. Lack of any one of these limits growth. An implication of this is that growth can only be sustained long-term at a level that does not exceed the supply of base cations available, since nitrogen is abundant. Biomass harvest is only sustainable when the removal of nutrients through harvest does not exceed the supply of these same nutrients through the atmospheric deposition, chemical weathering of rocks, and fertilization supplied externally .

A second corollary to sustainability that we suggest is the pollution criterion used to define the critical load of a pollutant. We suggest:

The system is not long term sustainable if the critical load of one or more pollutants is exceeded or the input of a given pollutant that lead to the maximum acceptable adverse effect the resource base, the structure or the function of the ecosystem

3.3 Aspects of sustainability

3.3.1 Natural sustainability

Several criteria must be met in order for forestry to be sustainable. These can be derived formally from the definitions just presented. A forestry ecosystem should preferably be sustainable with respect to the following functions and properties (Sverdrup and Warfvinge 1988; Falkengren and Eriksson 1990; Falkengren and Tyler 1991; Warfvinge et al 1993; Kimmins 1997a, 1997b; Ehrlich 1988, Sverdrup and Rosen 1999; Comins 1997; Skogspolitiken inför 2000-talet; 1992):

1. Preservation of the biomass production capacity
2. Preservation of the self-organizing potential for biodiversity
3. Maintenance of the physical stability of the soil
 - (a) Carbon storage being positive or balanced in the long run
 - (b) Maintenance of structure-bearing complexes maintained
 - (c) There being no net non-natural soil erosion

4. Nutrient resources

- (a) There being no long-term depletion of adsorbed stores of base cations, which amounts to there being a balance between inputs and outputs
- (b) There being no long-term depletion of reservoirs of phosphorus, which amounts to there being a balance between inputs and outputs
- (c) The C/N ratio remaining constant, which implies the long-term nitrogen storage not being greater than the rate of long-term carbon storage.

5. Chemical conditions

- (a) There being no internal accumulation of heavy metals or of substances alien to nature
- (b) The acid-alkalinity status stays within bounds of the natural system type

The aim of the field studies was to test and demonstrate the effectiveness of various assessment tools and other methods developed in the SUFOR programme. The programme plans to make use of all of these in due time. Since we do not yet have the capacity to test all of these tools and methods we concentrated upon the investigation of a limited number of conditions needed for sustainability:

1. Preservation of the biomass production capacity
2. Preservation of the biodiversity stock and the self-organizing potential
3. No long term depletion of the adsorbed stores of base cations, which amounts to a balance between inputs and outputs
4. The acid-alkalinity status remaining within bounds of the type of natural system involved
5. A momentary carbon balance at the Asa Forest Research Park
6. A momentary nitrogen balance at the Asa park

These matters were investigated on the stand level (10 ha) at our special study site at Jämjö, at the landscape level at Asa Forest Research Park (3,200 ha) and as applied to the whole country using information from the National Forest Inventory (22,700,000 ha). In terms of its being a productive unit in the human economical system the forest ecosystem must also be economically balanced. Under conditions of intensive farming of forest trees, the sustainability criteria must be met for a production cycle as a whole. Violating the criteria ultimately leads in most cases to malfunctioning or collapse of the ecosystem. This means that a site should always have a stable vegetation cover or potential for developing one. Thus, after harvesting, the site should be able to regenerate a vegetation stand just as productive and self-stabilizing as before, stable in the long run with respect to both large trees and to ground vegetation. This does not preclude the evolution of the vegetation and of the succession of species, but simply ensures that the development over time remains within a stable convolute, i. e. possesses convergent states. It implies harvesting from the site not exceeding the natural resources available. If no longer used for production and harvesting of biomass, it should be self-organizing in returning to a stable forest ecosystem of the same type as was there initially.

Forests can be used for several purposes such as for the production of biomass and timber, for recreation purposes, for the destruction of airborne pollutants, for hunting, for sequestering carbon dioxide from the atmosphere, for producing oxygen for human and industrial consumption in combustion processes, for preventing soil erosion etc... These activities tax the resources, the functioning and the structure of the forest system. A forest is a good example of its making good sense to determine and work within the framework of natural and economic boundaries to sustainability. In terms of social and economic sustainability, the following criteria should be fulfilled:

1. Economic conditions

- (a) The possession of a forest property should be profitable from the owner's perspective. Property rights need also to be protected against violations of against rights to personal integrity, at the same time as stay owners meet the obligations to society which they have within the democratic system.
- (b) The management of the nation's forest properties should be profitable from a national perspective as far as all the values concerned are involved with due respect being paid to matters of ownership and of personal integrity.

2. Social conditions

- (a) The management of a forest should be compatible with the Swedish legal regulations concerning free access to forests by the public and should see to it that those societal values not included in formal property rights are maintained.
- (b) The economic system for the individual generation of income and the societal redistribution of wealth should be predictable long-term and should take into consideration the long lag-times involved in the biological responses
- (c) The forest management paradigm should be compatible with the aesthetic preferences of society
- (d) There should be a flexible feedback system between lawmaking, economic actors and the electorate
- (e) No legal regulation can be allowed to be retroactive.

3.3.2 Social sustainability

Taking account of the starting point is important. This can be illustrated by listing the social paradigms in Sweden that presently apply (Kulturmiljövärd i skogen 1992; Jaeger 1995; Azar et al 1996; Williams and Matheny 1995);

- 1. The Swedish public considers free access to all extensively used forest land to be a civil right
- 2. Forests are expected to be a host to hunting each year that is open to a broad segment of the population
- 3. The public demands that forests have an aesthetically appearance
- 4. Forests are expected to provide opportunities for jobs and the substrate for the forest industry

Such paradigms are not constant over time, and may well change significantly in the future. The definition of sustainability and the corollaries to it are not limited to forest ecosystems but are applicable to any type of ecosystem, even to the entire vegetation on earth. The second corollary also stipulates that pollutants should be allowed to accumulate in an ecosystem in such a way that the system is not adversely affected (the critical load). It implies that the use of resources be such that when exploitation comes to an end the ecosystem is able to self-organize the recreation of its natural function, structure and resource status. In terms of biodiversity it implies that while we use the ecosystem for our purposes, all the biodiversity need not be maintained, but when we have ended our use, then the land must be able to regrow it original biodiversity by itself, all within a reasonable time. This implies that biodiversity at a certain location may be allowed to disappear, if that biodiversity can be maintained in another part of the landscape in such a way that it can recolonize the area where it was lost. The path to forest sustainability is of importance. Remaining for long periods outside the sustainability area

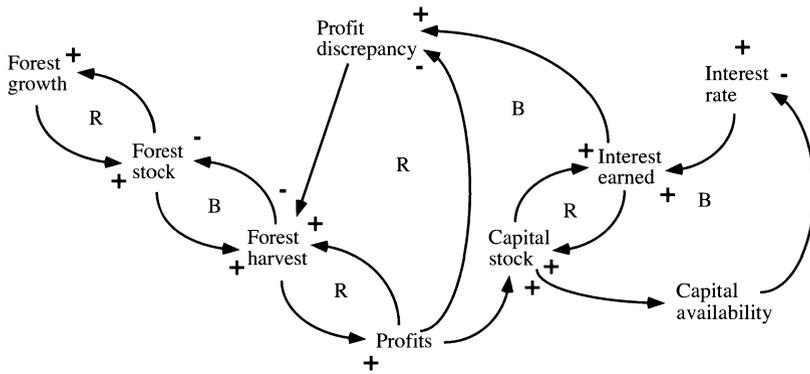


Figure 3.2: Under the present economic conditions, there are no constraints on the natural system of the transfer of values from it to the economic system unless the adverse effects of this become extreme, economic growth not being bounded by the actual capacity for sustainability, financial interest rates also being scarcely related to the economic profitability of the natural system.

may affect the sustainability limit, changing the position of that limit. The cumulative excess stress on the system caused by non-sustainability cannot be larger than the finite capacity of the resource reservoir used for transient resource borrowing. At the point where the resource reservoir is emptied, strict sustainability on the lower level will be thermodynamically enforced. In extreme cases the new low level may be zero, implying obliteration of the system. From the second definition the requirements for preserving biodiversity can be derived. A corollary would be the environmental pollution criteria that can be derived from the main definition: The system is not long term sustainable when the critical load of one or more pollutants are exceeded. The critical load is the input of a pollutant that lead to the maximum acceptable adverse effect on system resource base, structure and function.

3.3.3 Economic sustainability

The economic system is a vital component of any given society, it's also being a part of the social communication system. Although economic sustainability is an integral part of social sustainability, it can also be defined independently as follows (Skogspolitiken inför 2000-talet, 1992; Kimmins 1997):

1. The area of sustainability in any given economic system lies between the level of long-term balance and the level of net increase of production up to the limits of the natural sustainability and of the socially sustainable harvest fraction of the carrying capacity this represents.
2. For an economy to be sustainable the fluctuations that occur need to remain within reasonable limits and be long-term convergent within the area of stability within the response space.
3. Economic subsystems within the overall system are not sustainable, if the sum of all the economic subsystems contained within the boundaries of the system remain long-term sustainable.

The stability criterion states that regulative cycles need to take into account the slow feedbacks existing in the system, especially when these can result in adverse effects. A system is only

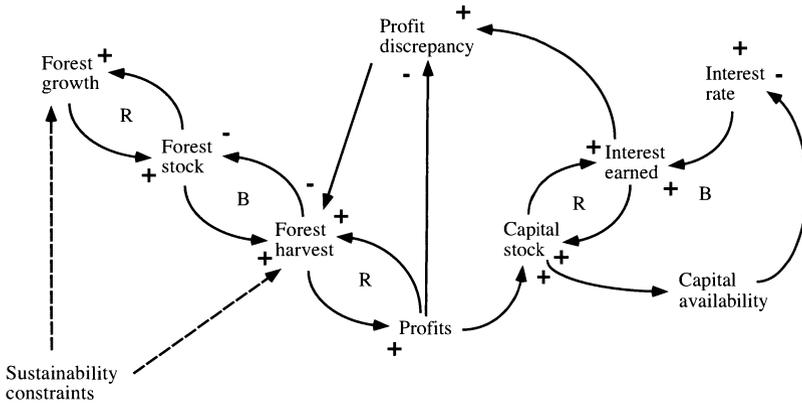


Figure 3.3: If the whole economy of a society consists of forest only, the financial interest rate in the society is obviously the ratio of profits to capital stock within the system. This makes it evident that the traditional interest calculus concerning the profitability of forestry operations within natural systems cannot use a financial interest rate that the system is unable to sustain on the basis of the carrying capacity it has. Since the natural sustainability constrains growth, harvesting is limited both by the sustainability constraints based on growth and by additional natural and societal limits.

long-term sustainable within the region where all the individual areas of sustainability overlap. Over long periods of time, areas outside the region of sustainability will become self-eliminating unless they are sustained by an excess of sustainability of one or more systems within the area of sustainability. In the present open economy, sustainability is in fact being traded between regions, even if this seldom occurs by way of purpose or design. Natural sustainability can be defined in terms of a mass-energy-time space; the coordinates of other forms of stability being the same, but significant difficulties needing to be overcome in order to express them in such terms. Economic systems are also under the control of the laws and incentives of the social system. Thus national economies are not sustainable if they are not stable and net long-term sustainable. If the government's budget in the matter is unbalanced, the imbalance needs to be made up for by some other unit or units within the government or by future mitigation on the government's part. National debt can be seen as a way of passing the question of a nation's sustainability capacity on to a future date. This is acceptable as long as it is pursued for long-term sustainability within the framework of a plan in which everything is accounted for in such a way that future generations do not have our problems thrust upon them. In the present open economy virtually no constraints of the natural system are transferred to the economic system, except when the adverse effects are extreme. The alternatives to forest activities considered within the system are based on conditions in the surrounding system (Fig. 3.2). Growth is not bounded by current capacities for sustainability capacities, interest rates scarcely being related to the real economic profitability of the natural system. As a thought experiment, consider an economy that in its entirety consists of the forest sector, in which all investment alternatives are basically identical forests, profits for all alternatives being the same. Adding sustainability limitations to such a system creates a distribution of profit rates, the average profit to capital-stock ratio being the interest rate within the system. It is evident that under such conditions interest rates are not independent of profits, since interest rates are derived directly from profits, though involving the distortions and signal noise found within society. Thus traditional interest calculus of the profitability of operations within natural systems cannot use a financial interest rate that on the average cannot be sustained by the system. Accordingly, natural sustainability constraints limit

available growth and social sustainability constrains in certain aspects the maximum harvest (Fig. 3.3), which is also subject to the constraints stemming from limitations to growth. Thus, natural sustainability limitations are transferred to profits through their setting a maximum on these, and in turn this limiting of profits to those that are sustainable, transferring to the interest rates on alternative investments. The present use of unbounded interest rates which are not coupled to the capacity for sustainability is convenient, but is also short-sighted and for the reasons taken up above, is fundamentally wrong. Thus, the economic system must be designed so as to include these limitations and the feedbacks that are missing. An alternative is to use the sustainable growth-rate-to-stock ratio as the physically determined interest rate of relevance to the system. Proposals for how this could be started will be presented.

3.3.4 Temporal dimensions

The time horizon to adopt in sustainability planning is one involving a very long time period, preferably forever. Sustainability in Sweden, for example, should thus be determined by first thinking about how long one envisions the Swedish realm to last. Do one want Sweden to remain in a state of prosperity for the next 1,000 years? If yes, the planning horizon must be one of at least 1,000 years. If shorter periods are chosen, for example 200 years, then this implies that we do not care what the consequences are after these 200 years have passed. The most usual justification given is "...we cannot make predictions more than 100 years ahead with any accuracy, and who knows, what kind of political system will be in charge then...". Taking for granted that in 200 years "someone smart" will come up with a solution to all problems that arise then from what we do now is just to find cheap excuses for not taking full responsibility for our own actions. Experience shows that sustainability needs to be planned in a sound way in terms of what we do and know today, without neither basing our continued existence on wonders occurring in the future nor ruining resources for those that come after us. The time perspectives for forest planning should be one of at least 250 years, and 500 - 1,000 years is needed for any serious sustainability perspective to be achieved. This can be set against the time-perspectives of global climatic changes and of measures against carbon dioxide pollution which are planned for a time horizon of 400-500 years. Although for acidification, the time perspective is one of mitigation within about 30 years, recovery is, in fact, estimated to take up to 500 years. For forestry, the only natural time unit for basing perspectives on is that of the length of one forest rotation. In Sweden this is from 70 to 120 years. A short-term economic perspective of 2-4 years in this context is completely pointless, its being a mere 2The following definitions of typical times involved in the ecological and societal processes is provided:

- **Market cycle perspective** 3-7 years is applied to investments and market evaluations
- **Carbon cycle perspective** 300-600 years, the time it takes to change the large reservoirs
- **Nitrogen cycle perspective** 100-200 years, the time it took to change the large reservoirs through pollution and management. This is strongly coupled to the carbon cycle
- **Birth, life and death of a tree** From 80 when trees are harvested to 240 years for a tree that is left as old growth
- **Acid pollution cycle perspective** ..is already 150 years under way. It took 100 years to acidify our soils and waters, and it will take an equal amount of time from 1980 onwards to recover from it.
- **Human population cycle perspective** is at least 100-200 years. Over the next 100 years, several predictions suggest that the worlds population will rise to something between 9,000 and 12,000 million people, and in the 100 years after that decline to a different level which is nearly impossible to predict.

- **Cultural heritage cycle perspective** is approximately 30-40 years, the time from father to son and the basic time unit for a significant cultural heritage change

This illustrates all the changes that may occur is the time from when we start our productive forest stands and until we can harvest from them. We can therefore also classify different time horizons:

Immediate time perspective The immediate perspective is one of 2-4 years, a mere blink of the eye within the time framework of a forest. This is the normal time horizon in traditional economics.

short-term time perspective A short-term perspective in forestry is some small fraction of a rotation period which lasts for 70-120 years, such as 10-30 years

intermediate time perspective An intermediate perspective is the length of a single rotation period, some 70-120 years.

longer time perspective A longer time perspective is one involving more than one rotation period up to three rotation periods, some 120 to 360 years. This is the shortest time period for considering the dynamics of nutrient, carbon and climatic cycles

perspective of sustainability Sustainability is a permanent perspective, where no time limiting being set.

When looking forward a hundred years it may be useful to look back. The difference from year 1700 to the year 1800 was very large. The change that occurred from year 1800 to the year 1900 was even larger. The change from year 1900 to the year 2000 was colossal. The experience of this tell us that the change that will occur from year 2000 to the year 2100 can be expected to be very large. The time perspective involved is important because of the effects of non-sustainability on the limits involved in sustainability. The cumulative excess stress on the system caused by non-sustainability cannot be larger than the finite capacity of the resource reservoir used for sustaining the system in a non-sustainable state. At the point at which the resource reservoir is empty, a lower level of strict sustainability on the lower level will be immediately enforced by basic thermodynamic principles. In extreme cases the new levels of sustainability level may be zero, implying the obliteration of the system. At this point are any complaints by the inhabitants completely ignored since no alternatives are available. An obvious example is that of agricultural practices that involve soil erosion and loss of soil into the sea. At the point at which no soil is left, agriculture can no longer be sustained.

3.3.5 The spatial dimensions

There are many examples in which changing the spatial scale over which sustainability is assessed change the conclusions one can draw about whether good stewardship is being practiced. Clear cutting of an area can remove all the timber in an area during a short period of time and eliminate all possibilities for the logging of timber in an area during a half-century or more. Although this is unsuitable at a local level, the rate of harvesting involved may be sustainable if a larger spatial scale is considered. This dimension of the sustainability problem is one that is more prevalent nowadays than it was earlier. Societies that have battled with such problems throughout their history have been operating on a much smaller scale than in the case today. The spatial dimension has also been used as an excuse for "business as usual" attitudes, since everything is so interconnected and so large, and thus so complex, leading to such statements as "we cannot do anything about it". Although the problem may be complex, the solution to many individual aspects of the problem may nevertheless be very simple. Interestingly, in the report "Our Common Future", the document that paved the way for the term sustainable development and that generated so much attention being directed at the issue of sustainability, one hardly finds any discussion of any types of sustainability other than the global one. It may also be that sustainability at one level of the overall system may counteract sustainability at another.

3.3.6 Biodiversity

The definition of sustainable biodiversity in all its dimensions and different aspects is still evolving, the field changing constantly as new knowledge emerges (Dallmeier and Comiskey 1998). How sustainable biodiversity should be defined is still being developed (Dallmeier and Comiskey 1998). Biodiversity sustainability is partly warranted by ecological and thermodynamic reasons (certain components may be essential ecosystem parts, maybe important for stability or a certain function), but also because of social and ethical concerns (A responsibility towards future generations, the avoidance of risks, ethical considerations on what humans may do or should not do). Biodiversity can be conceived as having dimensions of; number of species, b) amount of genetic variation, c) geographical distribution of a) and b) at present and as in terms of future potential, and d) development of diversity over the evolutionary time scale (White and Nekola 1992). The definition of the level of quality of biodiversity is not yet fully developed, and further development is recognized as being of major importance. Natural ecosystems are dynamic, biodiversity is subject to natural variations and evolve over time. Loss of biodiversity is replaced in the long run by newly created biodiversity. Although many of the processes are known, methods for conducting a full systems analysis that can yield objective functions and defined limits for sustainability are still lacking.

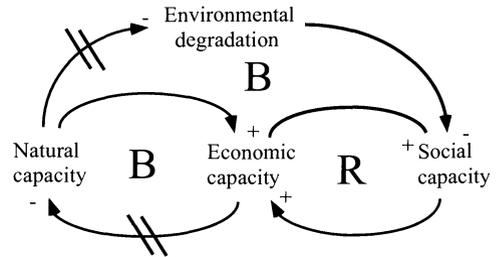


Figure 3.4: Causal loop diagram for the interaction between natural, economic and social capacity. Overuse of the natural resources by the economic activity will feedback negatively on the economy as well as negatively on society through environmental degradation and by declining economy. There is a problem in seeing the feedbacks from within the system because of the delays between economic expansion and natural resource depletion and the delay between natural resource depletion and environmental degradation. The delays are indicated with crossbars.

3.3.7 Harvest sustainability

The key operational definition is Liebig's law, which can in fact be derived from the law of mass conservation. It applies to any type of harvest taken from the land or sea, be it plant or animal. Harvest is not long term sustainable beyond the availability of the nutrient or source of energy in least supply in the production system. The term nutrient is to be understood as applying to all nutrients including water and gases. For plants, solar and abiotic chemical energy can also be seen as nutrients. The population corollary to sustainability can be derived from the Liebig's law as given above: No population of living creatures is long-term sustainable to a greater extent than the availability of the essential nutrient or energy type in least supply permits. Human being are an integrated part of all ecosystems they inhabit, and all their activities fall under the sustainability considerations made for that system.

3.3.8 The dynamics of sustainability

The limits to sustainability are dynamic and changeable, depending on the design of the economy and of social systems. Natural sustainability is the least flexible is of all types. Careful optimization can enlarge the area of total sustainability. A corollary to the second definition is that there needs to be direct connection between any executive power and responsibility for the effect of actions. This can be used to show that stable rules and conditions need to apply to private ownership for responsibility to prevail. Laws and incentives for profit also serve to control the economies of the world. One conclusion to be drawn from the definitions given above

is that national economies are not sustainable if they are not balanced in the long-term. The imbalance of one economy can only be endured if some other economy agrees to take over the losses caused by the unsustainability of the first mentioned economy, provided that its commitment does not exceed its' own limits of sustainability. A non-sustainable society may thus be sustainable if it can collect unused sustainability potential elsewhere. An economic market system provides the most efficient tool available for transferring unused sustainability potential between subsystems.

3.4 Integrated sustainability

The system as a whole is only long-term sustainable within the area in which all the component areas of sustainability overlap. Thus, the limits for each component needs to be determined quantitatively in order to determine the common area of sustainability. A system may be able to remain non-sustainable for a considerable period of time, bridging over its deficit by "borrowing" finite resources available elsewhere. Natural limits to capacity will soon set in, however, forcing the system to stay within the limitations set. The presence of non-sustainability for an extended period of time can erode the capacity for natural sustainability in such a way that the system moves from a position in which total sustainability is possible to one in which total sustainability become impossible because of the boundaries of natural sustainability no longer overlapping with the boundaries for sustainable economic or social activity, which implies the three areas of sustainability no longer having any common area which means that catastrophic events could be expected to occur. Such a system has passed the point of no return, either its' collapse being virtually certain, or requiring extremely drastic measures to correct the situation. The challenge is then to transpose the coordinates of the social sustainability space and the economic sustainability space to the same coordinates as the natural sustainability space, so as to create overlap and define the outer borders of sustainability space. This means transferring these three forms of sustainability to coordinates of mass, energy, entropy, space and time within the system boundaries that are set. Natural sustainability can be defined in terms of the physical and thermodynamic limitations to which all substances in the world all must conform, those of the conservation of mass and of energy. Mass is always conserved for all practical worldly purposes, and, in the long run, mass sinks are forced to be balanced by sources of mass. The principle is universal and enforces itself. As the theoretical physicist Sir Arthur Eddington once stated, "If a theory is found to violate the second law of thermodynamics, then all hope for it is lost, it must with certainty be abandoned in greatest humiliation". It should be recalled that human beings and civilisations are part of the ecosystems in which they live. From this, it follows that the activities of human beings fall under the rules of sustainability. The definitions that have been given above, it follows that sources of human beings (immigration, birth) must at least be equal to the sink in human beings (mortality, emigration), sometimes referred to as the Malthusian philosophy or Lotka-Volterra dynamics. It is important to fully realize, however, that that these principles are derived from the law of mass conservation in such a way that any political, religious or ethical desire for exceptions are always effectively overruled in the end. There is no escape from this principle. Ignoring it leads to population catastrophes in the mathematical sense of the word, the system involved being reorganized and drastic changes occurring. This is well known and has been documented for many animal and plant populations, the principles apply to an equal degree to human populations. Human beings are concerned about the preservation of personal integrity, yet in order for such values to prevail, remaining within the limits of long-term sustainability of the population is a necessity. If the needs of the populations exceed the resources that are available, the sink will exceed the sources by an amount determined by the amount of resources that can be borrowed for a limited amount of time. The standard of consumption selected is directly connected with the size of the populations that can be sustained, and it is an inherent property of the system that the growth of a population over the sustainability limits of the land in question can only be accomplished

through a lowering of the standard of consumption by an equal amount. The presence of non-sustainability for an extended period of time can erode the capacity for the natural sustainability in such a way that the system moves from a position in which total sustainability is possible to one in which where sustainability become impossible. We may summarize the severity of the derived rules as follows:

1. The limits of natural sustainability are unclear and difficult to determine, the consequences follows the thermodynamic principles which are absolute and self-enforcing. Breaking such rules have severe consequences if it implies moving beyond the stored reserves of the system.
2. The limits of social sustainability are malleable and soft. By changes in cultural heritage, practices now seen as unsustainable, may be made to be comprehended as being unsustainable. They are also subject to unconditional overruling by the limits of the natural sustainability.
3. The limits of economic sustainability are hard with respect to natural sustainability and subject to mass balance principles, but soft with respect to social sustainability.

Forests are used for a variety of purposes; production of biomass and timber, recreation, hunting, destruction of airborne pollutants, sequestering carbon dioxide from the atmosphere, producing oxygen for human and industrial consumption, and preventing soil erosion. These activities consume the resources, and use the functions and the structure of the forest system. Forests are a good example that it makes good sense to determine the natural and economic boundaries for the sustainability. The definition of sustainability given above and its corollaries are not limited to forest ecosystems, but are applicable to any type of ecosystem, even to the whole ecosystem of earth itself including the world's population of human beings. The second corollary to the definition also stipulated that pollutants should only be allowed to accumulate in an ecosystem in such a way that the system is not adversely affected (The critical load). This implies that use of the ecosystem will be such that when exploitation comes to an end, the ecosystem is able to itself organize the re-creation of its natural functions, structure and resource status. It is very much a matter of debate how much one should permit artificial reconstruction of the ecosystem to take place, the question being whether artificial reconstruction really does re-create original, self-regulatory systems. Regarding biodiversity, it is not essential that all the biodiversity there be maintained, when we exploit an ecosystem, yet when we cease to exploit it, the it should be able in itself to reestablish the original biodiversity it had and to do so within a reasonable period of time. Several criteria that need to be met in order for a forest to be sustainable have been described above. Most of these are easier to meet if active forest management is practiced. Since forests become incorporated into the human economic ecosystem, they need to also be balanced economically. Under conditions of the intensive farming of forest trees, sustainability criteria need to be adhered to, either during production or after a production cycle. Violating these criteria tends sooner or later to result in malfunctioning and collapse of the ecosystem. After being harvested a forest should with whatever help of management that is needed, should be able to regrow an equally productive stand of vegetation in terms of biomass as before. It implies that harvest taken from a site should not exceed the natural resources available at the site. If a site is no longer used the production and harvesting of biomass, it should be self-organizing in the sense of returning to a stable forest ecosystems of the same type as was there initially. One can regard the definitions of sustainability given above representing definitions of goals that are necessary. The time needed to reach these goals may be long and the path difficult. We will often discover, however, that there are a number of different goals of very considerable interest that can realistically be reached if we are willing to sacrifice other goals. Giving different goals different priorities and through assessment of the risks involved in not meeting certain goals are thus highly important.

Chapter 4

Principles of sustainable forest management

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"The soil gives a larger yield when one changes the species of tree seeds, a thing I am convinced about through my own observations. The soil also gives a larger yield when different species adapted to one another are allowed to grow together, however, one of them ought to belong to the group of deeply rooted and the other to shallow rooted species, for example: Oak with Beech, Ash or Elm; Beech with Scots Pine; Pine with Larch and so forth" (Israel af Ström 1822)

4.1 Introduction

The Royal Forester A. D. Israel Ström made the statement cited above in his 1822 book "Förslag till en Förbättrad Skogshushållning i Sverige" (A Proposal for Improved Forest Husbandry in Sweden), which was the main textbook on the subject in Sweden for almost a century. He was uncritically in favour of systematic clear-felling and the development of single aged stands. Israel Ström and his followers taught and defended clear-cutting in an era in which unregulated forest use was common. They rejected the method of selective use and of tree and branch harvesting practiced in Germany and in France. During the period since 1800 a number of different periods in south Swedish forestry can be identified, before 1820 natural householding was practiced, where forest were not industrially used in any significant degree, except in the mining areas in Bergslagen. During the period of 1820-1890, the influence of the Royal forester Israel af Ström was felt, leading to ordered clear-cutting systems, and the development of forest yield tables, production planning and growth promotion systems. In the period 1890-1910 much open land was replanted, due to a perceived lack of forests. During 1905 the first Swedish forest management law was enacted, stipulating the requirements placed on forest regeneration (Skogen-Sveriges Nationalatlas 1990).

During the period from 1940 to 1990 a modern industrial paradigm developed, with the strict promotion of Norway spruce monocultures and of strict clear-cutting (Norske Skog, 1990-2000, StoraEnso 1990-2000). This paradigm was dominated by a philosophy of large corporations adapted to conditions of northern Sweden, its influencing laws, taxation rules and governmental policies for half a century. The reaction came in the 1990's with the rise of the environmental paradigm, which promoted pluralism, abandoned centralistic planning tendencies and resulted in a general liberalization of all economic activity. In 1992, the Rio Conference took the step to establishing an agenda for achieving a sustainable world, the Agenda 21 (Agenda 21 i Sverige 1997). In a forty chapter report, the key challenges on a global, national and locale level were

described (IPCC 2000). A vision was created for the way forward. This set a large number of forces into motion, and had clear ramifications for forestry (Salwasser et al 1993; Mladenoff and Pastor 1993; den Elzen 1994). Sustainability forestry paradigms started to develop from this (Sandberg and Sörlin 1998). The present chapter outlines ideas concerning how principles of forest management could be changed in order to make forestry more sustainable. The main points in our message are as follows:

- Adopting a longer time perspective
- Including hidden assets and liabilities
- Including new and hidden costs
- Including new and hidden items of income
- Considering a hierarchy of multiple goals in planning
- Full transparency in all decisions

During 2000, forest stem growth was approximately 100 million cubic meters in Sweden. From a base cation nutrient perspective, sustainable harvest of approximately 80 million cubic meter would have been possible, the largest part in southern Sweden. Running growth is at present larger than what can be sustainably harvested. Of the present growth, 70 million cubic meter of stemwood were harvested in the year of 2000, part of this being whole tree harvest and slash collection for biofuel. There is at present a total standing stock of 3,100 million cubic meters in Sweden (Skogen-Sveriges Nationalatlas 1990), it increases by approximately 20-25 million cubic meters each year. The forest growth rate corresponds to an annual average growth rate of 3.3%, in southern Sweden, the annual growth is somewhat larger, 4.2% on the average. The forests in southern Sweden are largely privately owned, although certain aspects (biodiversity, access, mining rights, environmental state) are owned collectively and in the responsibility of the public collective (SOU 76:1992; SOU 1997:105).

Sustainable management differs from traditional management in applying a number of new measures of success as well as additional boundary conditions. There has been far too little holistic thinking in forest management development. There has also been a preoccupation with single production issues to the detriment of attending systems as a whole and to systems dynamics. The Swedish parliament has decided that all economic activity within the nation must be conducted from the standpoint of sustainability. According to the new Swedish forestry law applicable to all forest management, natural conservation issues are to be given as much weight as matters of production. This involves a large mental step forward from current practices, and will require considerable rethinking of the forestry paradigm that presently prevails, a process that has already started (Sandberg and Sörlin 1998). Single-mindedly maximizing harvest is opposed to sustainable management. The forestry establishment needs to internalize this and understand it thoroughly (Franklin 1993; Drengson and Taylor, 1997; SOU 1997:105). Swedish authorities are placing emphasis on forestry management principles, foresters and forest companies are expected to comply and to duly perform. This implies several things: a major change in basic management paradigm, a need for generating new knowledge and the defining of multiple objectives that any sustainable forestry management plan must meet (Salwasser et al. 1993). From now on, forestry practitioners have a variety of responsibilities: for production, for biodiversity, for the landscape, for the soil nutrients, the cultural remains and monuments, the public access and the environmental impact of the forest on the surroundings. The primary objective of SUFOR has been to develop tools for the prediction of sustainable forest production, for impacts of management, pollution and climate change that are environmentally sustainable and for optimizing economic yield within the constraints of the environment and of sustainability. Computer-based mathematical models are combined with process-oriented tools and empirical algorithms. The mathematical models will include simulations of the effects of silvicultural

methods, pollutants, climatic changes on ecosystem state. Different model tools are being adopted to all the scales necessary for the needs of the individual forest owner including regional perspectives:

- Single forest stands (0.1-100 Ha)
- Forest land properties or catchment regions (1-30 km²)
- Larger political regions (30-150,000 km²)

Our assessment of the stands at Jämjö, Blekinge Province, represents the first level, the assessment of the forest property in Asa Research Park, Kronoberg Province, represent the second level, and our assessments using the Swedish Forest Inventory represents the third level. The model system in particular describe the effect of management, climate and pollution changes on productive growth, yield, nutrient cycling, considering factors such as:

- Yield aspects and sustainability constraints to production with respect to
 - Methods of production
 - Tree species mixture
 - Site properties
 - Intentional landscape manipulation
 - Pollution
 - Climatic variability
 - Pest and disease related effects
- Feedbacks between biogeochemistry and profitability
- Feedbacks between biodiversity and profitability

The vision has been to work with a pluralistic forestry where many different components are brought together and where monocultures and mixed cultures are blended in a way that the system as a whole is sustainable. The ecosystem resources will be used to an extent where the use can continue for an unlimited amount of time. Some monocultures will have to be converted to mixed cultures, some may remain, other mixed cultures may be changed to monocultures. All are available components of a system that sometimes will have to find a new organization in order to stay within the limits of sustainability. A new and more broad-minded attitude will be of great help in this aspects (Peters 1991, Bossel 1998; Franklin 1993; Grönare Skog 1999).

4.2 Mixed-species stands as a component

Trees normally grow in mixtures. Under natural conditions, monocultures are exceptions that mostly are found only in areas such as the northern taiga, with harsh climate. The most common reasons within forestry for actively growing two or more species in mixed-species stands are the nursing of given species, reducing risks, and of biological conservation (Kely, 1992). In Sweden nursing has been the prevailing reason, the use of birch as shelter-wood in spruce regenerations. The major reason behind the limited use of mixed-species stands has been economic, due partly to the limited market for hardwoods. In conventional forestry practice use of monocultures have often been viewed as an efficient and rational method of producing bulk volume. Mixed-species stands have been regarded as more labour-intensive and to have lower production. The latter is probably due to the yield/ha of the species with the highest growth rate when grown in monoculture is expected to decrease in a mixture. In many cases monocultures

do yield more net biomass harvest per area and time unit (Kimmins 1997). There are many examples of equal or even higher total production in mixed-species stands (Carlyle and Malcolm, 1986; Brown, 1992; Man and Lieffers, 1999), especially when canopies are stratified (Kelty, 1988; Tham, 1988; Mård, 1996). In fact, a mixture of ecologically compatible species should theoretically yield more than a monoculture because of a more efficient utilization of above and below ground resources through differences in niche characters (Kelty, 1992; Man and Lieffers, 1999). When considering the use of mixed-species stands as a possible management procedure to achieve sustainable production of wood, the long-term nutrient supply and balance of the trees is important. A balanced nutrient supply is a prerequisite for tree vitality and sustainable production. In southern Sweden, Norway spruce monocultures show increasing signs of nutrient imbalance and soil acidification is progressing (Nihlgård et al. 1999, Falkengren-Grerup, 1993; Hallbäck and Tham 1985). Growing Norway spruce in mixed-species stands may be seen as a countermeasure to the nutrient imbalance problems. This is partly utilized in the "Kronoberg-method" where Norway spruce is grown under a birch stand for the first 20-25 years or in the "Ståål-method" where oak is driven forth under an earlier established stand of Norway spruce, where the Norway spruce is gradually removed during 60 years, yielding an early cash flow. These methods possibly have better nutrient supply, they consistently yield a better stem to branch ratio with a longer and straighter stem without branches, it has also been suggested that they suffer less from frost damage. Many of the postulated positive effects on Norway spruce nutrition in mixtures may be true for the other species in mixed stands. The reasoning behind the claim that mixtures of spruce and deciduous species will be more biogeochemically sustainable than spruce monocultures basically follows four lines:

1. Litter quality is higher in spruce/deciduous mixtures than in conifer stands, causing nutrient cycling to be faster.
2. The intraspecific rooting volume is possibly greater in a mixture of species of differing rooting patterns, such as Norway spruce and birch or oak, than in a culture with only shallow rooted species such as conifer monocultures.
3. Canopy filtration of aerosols (also including any pollutants) is greater in a spruce monoculture than in a mixed-species stand. This is an advantage in a clean environment, but turns to a disadvantage in a polluted environment.
4. Growing shallow-rooted species of high growth rate in monocultures can result in nutrient depletion of top soil layers. This is an advantage under conditions where nitrogen is strongly limited, but turns to a disadvantage when nitrogen is supplied in excess.
5. Multi-species plant communities generally handle attacks of pests and pathogens with less ecosystem structural damages and better physical stand stability.

Although the use of mixed-species stands can be shown to be a possible route to improve biogeochemical sustainability, as compared to conifer monocultures, system dynamics in tree species mixtures such as mineralization, root stratification, and nutrient uptake needs further attention. Much more work can be done to improve the understanding of which tree species combinations will be ecologically compatible, appropriate corresponding management schemes to achieve sustainability from biogeochemical, economical and biological conservation point of view (Almgren et al 1984; Braadland and Rognerud 1993; Mladenoff and Pastor 1993; de Jong et al 1999). The interaction between tree species in the stand and the soil and between different tree species in a mixed stand are important.

4.3 Landscape and forests

Large scale forestry has had significant effects on the Swedish landscape. For large parts of the landscape, the vegetation cover has been changed completely by human action. In the

southernmost province of Skåne, the land was cleared from dense oak-beech forests 5,000 years ago. By 1840, before the large emigration of poor Swedish farmers to America, forest cover in southern Sweden had been reduced to only 60% (4-5 million hectares), of what it is today (7 million hectares). 95% of the landscape in southern Sweden today must be characterized as a managed cultural landscape. The present appearance of the landscape, the tree species, the age structure of the forest, the physical structure of the stands, all is created by active management. The fraction of natural wilderness in a strict sense is vanishingly small in southern Sweden. More than 50% of the forested area had a complete change of dominant tree species. Approximately 25% of the total area has been drained artificially during the last 150 years. This probably released large stores of carbon in waterlogged soils (possibly somewhere between 2,000 and 5,500 million tons of C from 1870 to 1970). At a sequestering rate of 3-10 million tons of C per year in Southern Sweden, this will take approximately 200-1,000 years to sequester back from the atmosphere. In addition, the draining of the large areas have changed the hydrological properties of the landscape. We have faster hydrological response to precipitation events today, and lower water holding capacity in the forest soils. In hilly and mountainous terrain, soil stability and landscape hydrology is much affected by tree species, forest density and ground vegetation rooting structures. Such landscapes are sensitive to changes in the forest cover. No clear cases of adverse effects from forestry misjudgments or landscape mismanagement are known from Sweden, but several cases of soil flux and changed flood-patterns may possibly have been influenced by effects of forestry. For other countries, such effects are known from British Columbia, France, Switzerland, Austria or northwestern United States (Kimmins 1995; M'Gonigle and Parfitt 1998).

4.4 A time of changes

4.4.1 Global climate

The world is changing in ways never seen before. According to the Intergovernmental Panel on Climatic Change (IPCC 1993, 2000), a basic climatic change is certain to come. There will be many arguments about where and how much, but there is a resounding consensus that a significant change will occur and that change has already started. For southern Sweden the predictions converge on a 2-4 degrees centigrade increase over the next 100 to 150 years and an increase of rainfall in the west and decrease in the east (Alexanderson and Dahlström 1992; den Elzen 1994, IPCC 2000). This could potentially cause large amounts of carbon to be released from forest soils. It would cause potential forest growth to increase by 15-20% from an annual average growth of 4.2% to 4.7% in Southern Sweden, provided there is enough water, and an increase in the weathering rate by 10-25 %, provided soil moisture does not change. A decrease in soil moisture by 10% would be able to reduce both growth and weathering by 20-25%, cancelling any effect of increased temperature on growth. Several areas in Southeastern Sweden are already limited by water supply during the summer.

4.4.2 Pollution climate

The soils down to 1.5 meter soil depth have gone through a significant chemical change since 1900 caused by deposition of sulphur and nitrogen, a process that will be partly reversed during the next one hundred years. Protocols have been signed in Europe which will reduce sulphur deposition by 75% and nitrogen to 50% as compared to 1988 deposition levels.

It would be highly unwise to ignore these changes in basic conditions for management and planning, which run the risk of becoming very expensive in the future (Bahn and Flenley 1994, Matisoff 1988, Fagan 1994, Sverdrup and Rosen 1998, Marchak 1995; Kimmins 1997). One of the big changes is that the conditions for deciduous species will improve significantly, without a parallel improvement for conifers. Norway spruce will have passed its temperature optimum as the summer temperatures rise above 15°C, the optimum of the temperature curve is considerably

higher for broad-leaved trees. This implies that the commercial advantage enjoyed by Norway spruce in conventional forestry may partly disappear. From this it follows that increased management competence in deciduous tree and broadleaf forest management will be valuable and necessary. The use of multiple properties opens the door to a wider range of possibilities for products, variation in biodiversity, a broader range of product qualities and the active use of intra-species synergies to increase and support sustainability as described later in this report. Some of the management strategies used for deciduous forests is to some degree already being implemented in Denmark and Central Europe.

4.4.3 The market

In terms of markets, the world will change more than ever before in the time corresponding to the next three forest rotations (250-360 years). During this time, the world's population is expected to reach a level between 9,000 and 12,000 million people, the population may possibly start to decrease after 2150. During the same time, oil production is expected to have passed its maximum in 2080-2100 and to be in steep decline after 2150. Hopefully, several countries in South America and Asia will probably have increased their standard of living significantly. All this points towards an enormous market for wood and paper products in the foreseeable long range future (Marchak 1995; Annual reports 1990-2000 of Stora Enso OY and Norske Skog ASA). The increase in population densities, a much narrower energy market and possibly dwindling forests in many parts of the world, will make the market more aggressive towards access to resource-based products and raw resources. With the decline of oil as a source of cheap polymer fiber, the importance of wood products will have potential to increase in the very long run. With the fluctuating, but persistent ascendancy of some South American and many Far East Asian countries into prosperity, the effect on the market can only be amplified. With the increase in oil prices with the emergent scarcity after 2050 (Norsk Hydro Annual reports 1990-2000), increased price of polymer raw materials may cause a shift to forestry products as the source of fiber raw materials. The world's coal reserves are very large, and synthetic crude oil from such deposits can be expected to replace a part of the lost oil production for some time during 2020 and 2080.

4.5 A future market outlook

The global markets for paper and sawmill products has risen steadily since 1945. The consumption in total volume has followed the socio-economic development, and consumption of products follow the development of Gross Domestic product. The large producers of forest products in Europe are Germany, Sweden, France and Finland. However, it is a trend that the countries with large production but with small domestic markets are dominant among the large exporters. In Europe, most countries consume more paper and wood products than they produce, and for Europe a whole, there is a net import. The same applies to a large producer like Canada, the domestic production is actually a bit smaller than the total consumption. Large producers like United States, Brazil, Japan, all with with large populations, are not necessarily likely to become larger exporters if their demographic development and their increase in GDP continues. The general impression from the most recent literature, is that forest clear-cutting and forest mining continues in large parts of the world. The forest are denuded at an impressive rate, and a timber shortage is a possible scenario, unless the practice is changed. At present, only New Zealand and some European countries have started significant movement towards a sustainable forestry. Other large countries like Canada or United States have good research in the area, but much remains to be done. Old growth of redwood and Douglas fir is still being large scale logged (Marchak 1997, Drengson and Taylor 1997, Sandberg and Sörlin 1998; Aplet et al. 1993; Kimmins 1996; M'Gonigle and Parit 1996). Some of the very optimistic growth estimates for countries like Brazil, for which it has been said it could grow the total world demand

| Region | Production | | | Consumption | | |
|---------------|--|--|--------------------------------------|--|--|-------------------------------------|
| | Roundwood million m ³ yr ⁻¹ | Sawn wood million m ³ yr ⁻¹ | Paper Pulp Mt yr ⁻¹ | Roundwood million m ³ yr ⁻¹ | Sawn wood million m ³ yr ⁻¹ | Paper Pulp M yr ⁻¹ |
| Europe | 284 | 82 | 114 | 300 | 91 | 117 |
| Russia | 274 | 75 | 4 | 258 | 70 | 8 |
| North America | 590 | 158 | 175 | 563 | 142 | 155 |
| South America | 106 | 27 | 20 | 100 | 26 | 20 |
| Africa | 58 | 8 | 5.5 | 54 | 10 | 5.5 |
| Asia | 284 | 90 | 91 | 325 | 105 | 104 |
| Oceania | 33 | 5 | 5 | 23 | 6 | 5 |
| World | 1,600 | 457 | 410 | 1,600 | 457 | 410 |

Table 4.1: *WORLD PRODUCTION AND CONSUMPTION, PART 1; Approximate production and consumption of forest products in the world based on statistics from 1992. Since then, consumption and production has gone up significantly (10-15%). Mt is million ton. Data from SOU 76:1992 and Marchak (1995)*

in 3% of its area, the predictions assume soil nutrients to be totally unlimited. Many forests in the tropics are located on highly weathered soils, characterized by low weathering rates and low soil stores of nutrients, the forests continuity depend on a nearly complete cycling of the nutrients. Large long term forest production is not all possible on such soils on the resources of weathering alone. It is highly questionable whether the proposed or present high harvest rates can be sustained in such systems without massive additions of nutrients. Without nutrient replacements, these soils will be reduced to extremely low-fertility soils of very low value for any agro-forestrial activity. The output from several of these countries are high at present, but forest regeneration is small in comparison, nutrient replacement strategies non-existent and the standing timber stock is quickly dwindling in a majority of these countries. These countries will in the short term continue to dominate the world market (the horizon is the next 10-20 years) and keep prices for roundwood relatively low. However, over a full rotation period of 70-120 years into the future, many of these warm climate forests will be gone, perhaps forever (Marchak 1995). That would in essence remove the competitive advantage they presently enjoy in a fairly brutal way. At the present rate of deforestation, these forests are largely a one-occasion mining operation. Unless dramatic changes in forestry philosophy occur in the next decade in these areas with a large scale move from mining to sustainable management, a steady increase in roundwood prices should be expected after 2020.

4.5.1 Paper and pulp market

Sweden has a large market share in world pulp and paper industry, much thanks to a well developed industry with good access to raw material at good prices. The development of front line paper processing technology was also important in the past, giving Sweden, as a small country, the necessary competitive edge. The best predictor for paper consumption is by far GDP alone (SOU 76:1992). The approximate paper use per person (kg person⁻¹yr⁻¹) in any country is fairly well predicted by the Gross Domestic Products (GDP) in US Dollars:

$$\text{Annual paper use} = 0.0108 \cdot \text{Gross Domestic Product} \quad (4.1)$$

The largest users in 2000 was United States with 280 kg person⁻¹yr⁻¹, followed by Scandinavia at 250 kg person⁻¹yr⁻¹, Japan 160 kg person⁻¹yr⁻¹, EU 150 kg person⁻¹yr⁻¹. Asia and Latin America consume a mere 25 kg person⁻¹yr⁻¹. Africa has a considerably lower consumption than the rest of the world. This reflects that during times of growth, these regions represents very

| Region | Roundwood mill m ³ yr ⁻¹ | Sawn wood mill m ³ yr ⁻¹ | Paper Pulp mill t yr ⁻¹ |
|------------------------------|---|---|--|
| Europe | -16 | -10 | -3 |
| Russia | 16 | 5 | -4 |
| North America | 27 | 16 | 20 |
| South America | 7 | 1 | - |
| Africa | 4 | -2 | - |
| Asia | -50 | -9 | -13 |
| Oceania | 12 | -1 | - |
| Swedish production potential | 12 | 3 | 6 |

Table 4.2: *WORLD PRODUCTION AND CONSUMPTION, PART 2; Approximate net fluxes of forest products in the world based on statistics from 1992 (SOU 76:1992; Marchak 1995). Since then, consumption in Europe, North America and Far East Asia has gone up significantly (15-20%).*

large new markets. In Asia, a slow long term increase in consumption can be expected to follow total GDP growth. For newspaper paper, a steady by slow growth can be also expected, where as white copy paper is expected to increase faster. The European pulp production amounted 40 million tons annually in 1992, the consumption was 46 million tons. Total world consumption in 2000 is estimated at approximately 180 million tons annually. The pulp production of Sweden is approximately 10 million tons annually. The paper and cardboard consumption amounted to approximately 250 million tons annually, of this 10 million ton are produced annually in Sweden (Marchak 1995). Paper and pulp production is 410 million tons yr⁻¹ of paper and pulp consumed, 40 million tons yr⁻¹ is traded as export, and the net export flux is 21 million tons yr⁻¹. Much of the North American export goes to Japan which is the largest buyer in the market. Japanese imports from other East Asian countries are large. At present the forest industry is characterized by increased concentration on larger production units in the paper and pulp industry. This implies an ongoing long term decrease in raw material prices, narrower profit margins, and fiercer competition between the great industrial corporations. A part of this increase in efficiency will also reach the consumer. Increased farming of fast growing poplars and pines in warmer climates will press market prices in the intermediate future. When the Russian economy is reorganized and starts growing, conifer timber exports may be expected to rise and remain high for a substantial time. Only in the very long term may a lack of pulping raw material be expected. Thus prices can be expected to stay fairly stable or slowly decrease.

4.5.2 Sawmill products

The prospects for increased profitability may lie in production of quality sawn wood, which traditionally always have had better prices. Many countries have good conditions for fast growing tree species, fewer have good conditions for quality timber. The plank and sawmill industry in the Nordic countries lag behind in this development. At the present time, 2001, the production units are small, under-invested and dispersed, and the market is characterized by poor profitability, and poor industrial logistics. Despite the fact that many tree species grow in Sweden, very little else than pine and spruce is converted to boards and building material. This is much a matter of attitude and failure to see and utilize market potentials. Oak and beech is used for quality flooring and furniture purposes, but it is completely unavailable for the general consumer for use in housing construction. Looking at other markets, this is an example

of a chance being overlooked. Other wood species are not commercially used in any significant scale. The Canadian market situation suggests that a market possibility present.

The consumption of sawed timber in Europe was 85 million cubic meter in 1986, of which 20 million was hardwood, imports to Europe accounted for approximately 20 million cubic meter (SOU 1992:76). The total roundwood production in Europe amounted to 284 million cubic meter annually, but consumption was 298 million cubic meter annually. A large fraction of this was produced in Sweden, Finland and Norway. In Sweden the domestic use has been 5 million cubic meter sawn wood (almost exclusively conifer wood) for a long time, it reflects population and GDP development. The production of sawmill products was 83 million cubic meter in 2000, the European consumption was approximately 130 million cubic meter, and 50 million cubic meter was imported. This represents an increase of 45 million cubic meter in total consumption of sawn wood. Of this total consumption, 100 million cubic meter was from softwood and 35 million cubic meter from hardwoods. The increase in the period 1986 to 2000 was stronger for hardwoods than in softwoods, and this development is expected to continue. Of the consumption, 60% is for building, 7% furniture, 23% for package and 12% is impregnated wood. The furniture fraction is increasing, and some hardwood has begun to be used in housing (Marchak 1995, Dregson and Tyler 1997). Of the total world roundwood consumption of 1,600 million m^3yr^{-1} , of this the export market is 125 million m^3yr^{-1} but the net export flux is 66 million m^3yr^{-1} . The sawn wood world production is 457 million m^3yr^{-1} of this, 87 million m^3yr^{-1} is traded as export, and the net export flux is 22 million m^3yr^{-1} .

Extrapolating consumption preferences from North America, and the past market development in Europe, business opportunities in sawn hardwood business, could be the fastest growing consumer segment in Europe. It could potentially be in the range of a total turnover of approximately 5,000-10,000 million SEK within a decade or two, and Sweden's share in this business could be substantial (SOU 76:1992, Marchak 1995). In the world at large, with dwindling hardwood resources in the tropical countries, a good market would be available for countries with standing stock of hardwood that could be harvested sustainably.

Our neighbouring countries in Europe, such as Poland, Germany and Ukraine, are large exporters of bulk wood and sawn hardwood products from oak, beech and ash. In these countries, hardwood cutting is at present larger than regrowth. In the International market Canada and several tropical countries are active, in the tropical countries, logging rates for hardwoods by far exceed the regrowth. At the present rate of forest destruction, these countries face a hardwood shortage within 20 years. In the future, there could be an increasing market share to be taken for a country with stable regrowth and quality timber to harvest. A restructuring of the sawmill industry, together with a development of a hardwood construction material market could improve the possibilities for hardwood forestry in Sweden.

4.5.3 Wood for energy

There is a relatively small but increasing market for wood for energy in Sweden, considering the total volume of energy generation in Sweden. The wood is used to a small extent in central heating units and mostly in the industry for heat. Furthermore, the use of wood for private house heating is small and the trade is to a large degree decentralized. Probably, Sweden's political reluctance over continued nuclear energy after 2010 will continue, and no or few new hydroelectric plants will be constructed, an increased political pressure for using more wood can be expected. The bioenergy sector is not very profitable, and imports of cheap coal-based energy loom as a permanent threat to the bioenergy sector for the next decades. The free market of EU precludes border restrictions. Much wood is derived from other forest industries as debris (chips and sawdust), as well as extensive collection of forest slash during stem harvest in Southern Sweden, the "whole tree practice". Considering the large nutrient removal with slash collection, this source will soon be burdened with nutrient return costs, losing further in competitiveness. Removal of slash removes significantly more nutrients per ton than stem harvest, and the whole tree harvest practice will under sustainability restrictions, imply a substantially lower possible

total biomass harvest than when focus is on maximizing biomass in stems. A more aggressive policy towards "no net CO₂ emission policies" may create future market potential for wood as an energy source. In the long perspective, wood for energy, which is an industrially low grade product, will have to compete with the demand for wood as a raw material for fiber, paper, pulp and sawn wood, having a substantially larger value generation in the market. De-regulation of the energy market in Europe have reduced energy-prices significantly, and they are expected to stay low for a considerable time. Thus, the future for forest fuel-wood in large quantities appear as weak when considered against the alternative uses and the projected prices for coal-based energy in the next decades.

4.6 Certification

4.6.1 Introduction

Environmental certification has become an important environmental quality tool for the forest industry. It implies that the forest owner agrees to manage the forest according to certain rules (Drengson and Taylor 1997; Grönare Skog, Swedish Forestry Board 1999). The certification is obtained after reoccurring inspections. Common to all the systems, is that the forest owner agrees to create a Green Plan for the management of his forest. Many consumers demand this as an insurance for quality. The certification implies to a large degree the introduction of planned management with a proper paper trail at all times, increasing process insight and documentation of all actions. However, the environmental content of the certification process remain incomplete and there will be an increasing demand from NGO's and consumer groups for more specific declaration of the environmental benefit. Increased awareness and scrutiny will increase the demand for the ability to document the sustainability of the forestry operation, where facts will win and rhetoric will loose. Sweden has always been a large power in terms of initiative, and have all possibilities to remain so. In is therefore important that the certification procedures are updated, and that the environmental content of the procedures steadily increased over time.

4.6.2 Certified management

ISO certification demands the management to have environmental steering systems, a developed environmental policy. The certified is required to follow the environmental laws and regulations as well as have a program to steadily improve the environmental compliance. In EU the standard EMAS is applied. In Sweden this has been used by larger forest industries and corporations, but the relevance and practicality is less for small private owners, much because of the cost and bureaucracy involved.

4.6.3 Forest Stewardship Councils; FSC and PEFC

The FSC is a global system for forestry certification, using 10 principles based on economic, social and environmental considerations. The basic principles were developed and discussed at a conference in 1993 in Toronto, Canada. The purpose of the certification procedure is to promote better sustainability of forestry. This provides a management vehicle for improving the sustainability requirements placed on individual forest owners. These have been developed into national standards, in Sweden this was determined 1998. Certification by Forest Stewardship Council gives the right to issue products with a FSC stamp. The Swedish forest owners association has chosen to develop certification models based only on a biological standard and have adapted them specifically to family-owned forestry. The owners are certified in groups and regular inspections are involved. These are national Swedish certifications, and offer a possibility for including more sustainability goals. This is called PanEuropean Forest Certification (PEFC).

4.6.4 Conclusion

The different certification standards available offer an excellent opportunity for standardizing procedures that promote sustainability. At present the requirements of the FSC, PEFC, ISO 14001 and EMAS are more environmentally oriented than focused strictly defining sustainability precisely. At present, the sustainability content in these standards are more of a nature of demonstration of good intentions and accountability. The quantitative sustainability requirements are almost always very obscure or absent. The potential for change is present. Mechanisms for steadily evaluating and developing these standards will be valuable tools for promoting sustainable forestry in a long term perspective.

4.7 The management planning procedure

4.7.1 Introduction

The management of the world's forests are at present largely unsustainable by a wide margin. A drive towards more sustainable forest management has developed in North America and Europe, a literature on the subject is arising, but practices remain at present mostly with business as usual. The forestry policy of several countries are largely focused on production, with little or no real concern for sustainability. Management methods in many countries such as Canada, United States, Indonesia and China must still be characterized as ecosystem mining, often removing the forest for a very long time to come. In this respect, the European countries have come furthest in approaching the concept of sustainable management. In a world perspective, our policy of compulsory replanting and regeneration appears as radical. Outside Europe, replanting or active regeneration occur only on a small fraction of the area (Marchak 1995). The forest management procedure needs to be modified to a more system dynamics/engineering type of planning method. The quality of management plans should always assessed with predictions of the outcome after implementation of the plan, in order to assess if the plan really can produce the intended.

It is important that the proposed plan is iterated several times and repeatedly tested against the criteria, in order to optimize the approach towards full sustainability. Fig. 4.1 shows this adaptive process. When conflict of criteria or interests occur, a structured model for handling these must be employed.

It is necessary to differentiate between what is best decided and planned at the local scale, and the planning that must be made and coordinated at the regional level in order to make sense (Bossel 1998). Especially for biodiversity, large scale coordination can be cost saving and

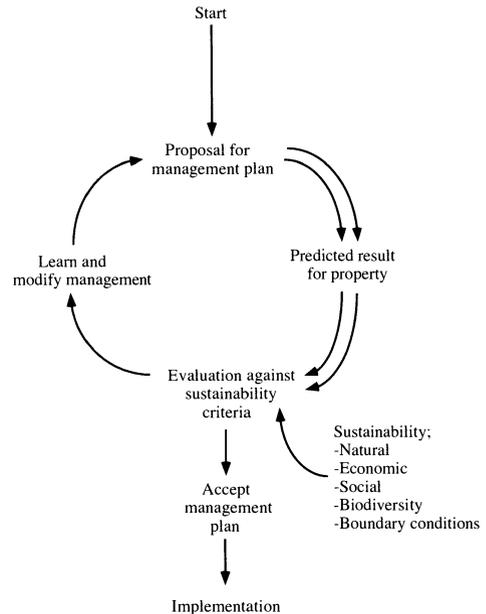


Figure 4.1: *The forest management procedure involved an iterative, adaptive method of designing the management plan. It is important that the proposed plan is iterated several times round the planning cycle and repeatedly tested against the criteria, in order to optimize the approach towards full sustainability and simultaneously profitability.*



Figure 4.2: *"It is obviously of great importance to choose tree species, management method and so forth that are well suited to the soil conditions. This always acknowledged principle have in many cases resulted in the use of a site class to assess the soil with regard to species choice, stand management etcetera. In this manner one has, by and large, managed to make correct decisions. However the site class concept becomes far to undefined for a finer characterization of soil conditions. A deeper understanding of soils is for this purpose of greater value as it can provide a possibility to assess whether the site class is suitable or whether it is in a state of change in one direction or another. A sharp characterization is central to the application of experiences regarding forest management at sites other than where these were originally made. An increased knowledge about soils is consequently conducive to increase the precision in the general management of forests."* The statement was written in 1928 by Professor Olof Tamm at Uppsala University (*Sveriges skogar*; Ed: A. Wahlgren and G. Schotte 1928). Then they pointed out the importance of soil resources and soil quality for the potential for growth and production in the forest. At that time the first soil maps were made and the first forest inventory attempted. The picture shows beech and alder forest from southern Sweden. If the predicted global climate change is realized, this will become the normal ecosystem type for a larger part of southern Sweden. Photograph by Peter Schlyter.

efficient. The criteria are tested in due order of what is considered to be the most important. When important criteria are violated, the consequence must be mapped and costed. Key to sustainable management is to define the objective functions. We can define the following sorting principles;

- Production goals
- Natural resource goals
- Cultural goals
- Social goals

It is important in all management planning that the time perspective is clearly stated. For sustainability a long perspective is required, and this must be coupled with the economic requirements which operate on both short and long time scales. The productive goals deal with the maximization of commercial production within the system constraints, utilizing the properties of the stands, utilization of the different properties of the different tree species and the capacities of the owner (Hägglund 1991; Grundbok för skogsägare 1985). The goals of natural conservation include the biodiversity goals, landscape structures, boundary conditions as defined by environmental laws and regulations.

Many forest estates in southern Sweden have highly developed cultural landscapes of long continuity and good aesthetic values. Important boundary conditions in Sweden are set by the quality of runoff to streams and lakes in the forest landscape. The soils are very sensitive to disturbances in the acid-alkalinity conditions in the catchment, as well as the degree of intensity in production. Many streams and lakes have fish and aquatic populations that are marginal or endangered by acidification and this requires special attention. The cultural conservation aspect takes into the account the historical aspects of the landscape and forest estate physical geography, the ancient remains, the landscape structure and the traces of past land use. Ancient remains are protected by Swedish law, and their destruction is punished. The social aspects involve hunting, access to the public for recreation, collection of berries and mushrooms, sports and education. The aesthetic component of the recreational aspect is considerable and considered by the

public electorate as important. By analyzing and sorting the assets of the property, the tree productive assets, the natural conservation assets, the cultural conservation assets and recreational assets, the totality of the property can be put into a balance sheet for assets, all this should be investigated and prepared as input data for sustainable planning. In order to do the analysis, data and information about the property needs to be collected and properly archived later. Very important in the assessment, is information about the most important resource for forest production, the soil. In present forestry, not much is available on this on the property level. For the agricultural sector, this is seen as essential. It is essential for planning of sustainable forest production, and in cases where this is missing, it should be sampled. It has been estimated in the SUFOR programme that the minimum cost for a nutrient planning with a 300 year perspective would cost approximately 5,000 to 10,000 SEK for a 50-100 hectare property. The average forest property in Götaland is actually 65 hectares. The data would for a large part be inherent soil properties with a valid date for centuries to come.

4.7.2 Planning structure

The analysis starts with assessing the three main sustainability dimensions and the boundary of the sustainability space. First the natural sustainability parameters are determined. This is done by estimating the production parameters from a perspective of sustainability:

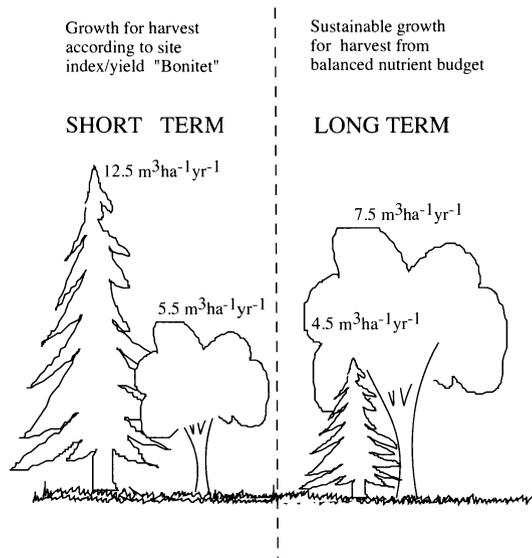


Figure 4.3: *It is necessary to rethink the forest yield system, "Bonitet". The forest yield expresses a measure for the maximum production capacity of the system. This is not the same as the optimal production rate. Thus the traditional yield, "bonitet" determined for Norway spruce, is generally larger than the sustainable yield. The traditional yield system classes deciduous trees lower than Norway spruce, but sustainable yield generally come out larger than traditional yield for deciduous trees, and substantially larger than the estimated sustainable yield for Norway spruce on the same site.*

1. Maximum possible forest yield, (Swedish "bonitet"); (short term perspective)
2. Sustainable forest yield (long term perspective)
3. Yield requiring external support (long term perspective)

which should be mapped on the property. The different types of forest production values have to be determined from traditional yield, "bonitet" to give the maximum production, from analyzed soil properties and used of the ForSAFE software to get sustainable yield. The yield requiring support is calculated as the difference between maximum production and sustainable yield, and also converted to amount of fertilizer per hectare and rotation. If the sustainable yield is larger than maximum yield, the "bonitet", then an increase in production above the traditional yield would be possible within the sustainability limits. These values define the yield that can be realized within the resources of the soil itself, the maximum stand growth capacity on the property and the required nutrient addition to make the maximum production possible. A selection of production level is made, as well as production type and product quality and these items are placed in the property. In this process, areas with lower or sub-profitable sustainable production capacity can be utilized for harbouring the elements for biodiversity and landscape values. When the aspect of production of quality is introduced, then mere bulk production becomes only one element of the assessment. The economic objective parameter would be income production per hectare, and then including all income and expense items. Thus, under certain circumstances, lower bulk volume of higher quality can be more profitable than high volume and low quality. It should be possible to derive volume-quality price curves by surveying the price situation in the relevant market. Next the boundaries of the economic performance of the property will be assessed, by performing an economic performance analysis under different tree species, stand localization and other parameters according to the economic model outlined below. Finally, the social restrictions by laws, regulations and public perception must be assessed, and the variability freedom is estimated. These three types of boundaries set the limits for operation of the forestry under sustainable conditions. It will be inevitable that there will be conflict of goals. The case may arise where the boundaries of sustainability for the three criteria do not overlap. Then the intended land-use is unsustainable, and a complete rethinking of the intentions is required. For many cases, a lowered production may well be a required measure to achieve sustainability. One way to achieve this is to allow longer rotation periods, sometimes also followed by higher wood quality. This will be accentuated as one progresses northward toward significantly colder climate. However, the amount of such areas are almost matched by areas presently classified as low productivity, where the carrying capacity is in reality high, and more production can safely be made. Thus, inclusion of optimization would be important for combating production losses caused by violation of system sustainability criteria.

In Tables 4.3 to 4.7 we have summarized a number of tree species properties which are important to consider in management and sustainability planning. Table 4.3 shows an overview of some of the economic properties of trees under management and effects of different factors on different tree species. The price indicated are very approximate and to be used for comparison purposes only. Assumed is a straight log of at least 6 meter length, no large branches is required for log quality A. Flooring is made as mostly composite boards but also to a smaller degree as whole tree boards. Oak, and ash are well suited for construction under many circumstances, but not much used anymore, especially oak has very good structural properties. Table 4.4 shows price structures, rotation period length and soil requirements. Table 4.5 shows reaction to environmental and climatic factors. The root penetration of the ground is always limited by the groundwater table, which only alder can penetrate. Table 4.6 shows further properties. Wood quality is related to the change in quality caused by increased growth rate. The rejuvenation properties are classed as emergent if the tree will emerge as the secondary tree taking over the stand of a pioneer. Weedy implies that it multiplies profusely. Weak implies that it competes

| Species | paper industry use | Construction use | Household use | Hardness | Price at sawmill |
|---------------------|--------------------|---------------------|---------------------|----------------|------------------|
| Spruce ¹ | paper and pulp | simple boards | firewood | straight, soft | 100 |
| Pine ² | paper and pulp | quality boards | furniture | straight, soft | 130 |
| Larch ³ | paper and pulp | quality boards | furniture | straight, soft | 130 |
| Birch ^{4a} | white paper | no | furniture, firewood | modestly hard | 200 |
| Birch ^{4b} | white paper | no | furniture, firewood | modestly hard | 200 |
| Alder ⁵ | white paper | no | matches, furniture | modestly hard | 100 |
| Aspen ⁶ | white paper | no | matches, firewood | soft | 100 |
| Cherry ⁷ | intolerant | flooring, veneer | furniture | modest hard | 300-750 |
| Oak ⁸ | - | flooring, veneer | furniture, firewood | hard | 300-750 |
| Beech ⁹ | white paper | flooring, veneer | furniture, firewood | brittle, hard | 200-500 |
| Ash ¹⁰ | white paper | flooring | | hard | 200-500 |
| Maple ¹¹ | white paper | flooring, ornaments | furniture, syrup | hard | 200-500 |

Table 4.3: *MANAGEMENT PROPERTIES; PART 1; Overview of some of the economic properties of trees under management in southern Sweden and effects of different factors on different tree species. The price indicated are very approximate of course and to be used for comparison purposes only. The price of Norway spruce pulpwood was set at 100 and all other have been made relative to this. The species are: ¹*Picea abies*, ²*Pinus sylvestris*, ³*Larix decidua hybr.*, ^{4a}*Betula verrucosa*, found on slopes and in the mountains and hills ^{4b}*Betula pendula*, the lowland variant, often on richer soils, ⁵*Alnus spp.*, ⁶*Populus tremuloides*, ⁷*Prunus cerasus*, ⁸*Quercus robur*, ⁹*Fagus sylvatica*, ¹⁰*Fraxinus excelsior*, ¹¹*Acer platanoides*, also favor soils with a little more water than normal.*

weakly with aggressive pioneers. Table 4.7 shows the price ratios between different tree species and timber qualities as they were in year 2000.

Finally, in several cases, goal conflicts and conflicts with boundary constraints which cannot be resolved on the local scale, can be solved in a larger landscape of regional scale. Then the active participation in landscape management by the small landowners will be important. Thus, a sustainable management regime among the landowners, will require a landscape-scale sustainable management organization. Experiences from Northwestern United States and from Canada, show that private or ideal non-for profit organizations can fill this purpose well, even if private forest ownership in these countries is underdeveloped (Marchak 1995). The ownership structure in a country is important. M'Gonigle and B. Parfitt, (1996) found out by studying data from Sweden and Canada, that small landowners have better management and appear to run their forests management much more sustainably than larger owners and far better than any corporate or state owned property. Overall, large corporations and government seems to be the worst for non-compliance with the principles of sustainability as well as production stability as compared to the smallest landowners. The small landowners are small, self-confident and independent with a high degree of integrity, and top-down steering applied to them may be very counterproductive. It is also important that this larger scale organization includes the necessary feedback mechanisms to entertain support from the stakeholders. The active participation of the landowners and the stakeholders will be important for accountability of all the members and the resolving of conflicts between them on all scales.

| Species | Required dbh | Rotation time years | Final stem density per hectare | Regeneration management | Soil |
|------------------------|--------------|---------------------|--------------------------------|-------------------------|----------------------|
| Spruce | 0.3 m | 60-70 | 250-400 | Natural | Poor |
| Pine | 0.4 m | 70 | 250-400 | Natural | Poor |
| Larch | 0.4 m | 70 | 250-400 | Planting | Alkaline |
| Birch | 0.4 m | 50 | 250-300 | Natural | Any |
| Alder | 0.3 m | 50 | 250-400 | Natural/Planting | Wetland |
| Aspen | 0.3 m | 40 | 250-400 | Natural/Seeding | Alkaline |
| Cherry | 0.3 m | 50 | 250-400 | Planting | Rich |
| Oak | 0.7 m | 120 | 50-100 | Natural/Planting/Boar | Rich, deep |
| Beech | 0.6 m | 110 | 150-230 | Planting/Natural/Boar | Rich, deep |
| Ash | 0.4 m | 80 | 150-200 | Planting | Rich, moist, drained |
| Maple | 0.5 m | 80 | 150-200 | Natural/Planting | Rich |
| Lime ¹² | 0.4 m | 80 | 200-250 | Planting | Rich |
| Hornbeam ¹³ | 0.4 m | 100 | 150-200 | Planting/Natural | Rich |

Table 4.4: *MANAGEMENT PROPERTIES, PART 2; Some management properties used for planning next tree generation. The species additional are: ¹⁰ Tilia cordata, ¹³ Carpinus betulus.*

4.7.3 Economic principles

Introduction

There are three different economic systems the forest management world can operate in (Bossel 1998). Most people working in the forest sector have been educated in the traditional paradigm, where large corporate management is the prevailing. Any occasional smaller private owner simply have to go along with that. Such are the conditions in Northern Sweden. In southern Sweden, the situation is significantly different. The forest is largely private owned by farmers, where the forest is a larger or smaller part of the farm economy. The paradigm that applies to these forest owners is very different, it is more affected by a combined farming view and the time perspective of the farmer significantly more long term. The typical farmer often has a full- or part-time job in the next town or city, runs his agricultural land part of the time and works with his forest some weeks every year. Very important for the smaller landowner is also the possibility to accumulate capital at a low fortune tax rate in order to be able to financially manage the next generation change at the farm (Seymour 1976; Wagner 1939, Kulturmiljovard i skogen 1994, Kuechli 1993, Clarke 1991). Though this is generally discounted by traditional industrial forest management as irrelevant, it occurs on a very substantial scale that is well documented through the tax records. This also illustrates the time perspective of several family generations envisioned by the small landowner.

Income and cost accounting

Several things will change in the future. Economic forecasting for smaller farms with forests in the size from 25 to 300 hectares must be much better adapted to the existing realities. It is recommended to take into account the income from the following items:

1. harvest
2. net change in standing stock of trees

| Species | Drought stress | Shadows tolerance | Nitrogen effect | Frost resist | Wind resist | Acidity tolerance | Nutrient capture |
|---------|----------------|-------------------|-----------------|--------------|-------------|-------------------|------------------|
| Spruce | intolerant | good | fast | good | poor | moderate | sufficient |
| Pine | very tolerant | modest | tiny | good | modest | moderate | intermediate |
| Larch | intolerant | poor | small | modest | solid | sensitive | intermediate |
| Birch | modest | poor | modest | poor | modest | resistant | intermediate |
| Birch | intolerant | poor | modest | good | solid | resistant | intermediate |
| Alder | intolerant | poor | small | poor | modest | very sensitive | surficial |
| Aspen | intolerant | poor | large | poor | modest | very sensitive | intermediate |
| Cherry | intolerant | poor | large | poor | solid | resistant | intermediate |
| Oak | modest | poor | small | poor | solid | resistant | deep |
| Beech | intolerant | modest | small | poor | modest | resistant | deep |
| Ash | intolerant | poor | large | poor | solid | sensitive | deep |

Table 4.5: *MANAGEMENT PROPERTIES, PART 3; Overview of some of the properties of trees under management and effects of different factors on different tree species. The table was synthesized from different general forestry handbooks, but considerably changed after discussions within SUFOR.*

3. hunting and fishing rights
4. increase in soil stocks of nutrients
5. contributions from recreational activities
6. other land-use in the forest estate

It is important to be realistic and develop an economic accounting model that corresponds to the actual timing of costs and discounting of items. The income items are determined by both traditional items, and some new items overlooked earlier and from complementary activities such as hunting or recreational activities. The harvest is booked at real sales price, the increase in stock at the same rate, hunting and fishing rights at their market annual sales value realized, the increase in soil nutrient stocks at market price for delivery to the soil of comparable fertilizer product. For recreational values and changes in cultural stocks, no robust pricing mechanisms are available at present. Some cultural values give subsidies from the government, these must be booked as income, and increase or loss of stock at the same rate, proportional to volume of the stock. A new accounting item is the income generated from weathering which is not used for harvest, which will show up as an increase in the nutrient storage in the soil. This storage belong to the property owner, and can be used at a later time.

The cost items are the traditional cost items, but with the addition of some new items, such as cost of breaching boundary conditions. The cost accounting needs to take into consideration the effective cost of:

1. nutrient addition
2. rejuvenation
3. stand management
4. annual depreciation for infrastructure investments
5. annual costs for infrastructure maintenance

| Soil type | Bulk of fine root distribution | Root rot | Fast growth wood quality | Animal browsing intensity | Rejuvenation strategy | Species |
|-----------------------|--------------------------------|----------|--------------------------|---------------------------|-----------------------|---------------------|
| moraine, peat | 0-0.4 m | much | poor | some | fast emergent | Spruce ¹ |
| moraine, sandy | 0-0.6 m | some | poor | much | slow emergent | Pine ² |
| loose, drained | 0-0.7 m | much | poor | much | weak emergent | Larch ³ |
| moraine | 0-0.6 m | some | good | much | fast pioneer | Birch ^{4a} |
| loose, drained | 0-0.7 m | some | good | much | fast pioneer | Birch ^{4b} |
| moist, organic | 0-0.4 m | much | - | much | weak pioneer | Alder ⁵ |
| drained | 0-0.5 m | much | - | much | weedy pioneer | Aspen ⁶ |
| moist, drained | 0-0.6 m | some | good | much | weak pioneer | Cherry ⁷ |
| clayey, heavy | 0-1.0 m | some | good | little | need fauna | Oak ⁸ |
| brown, drained | 0-0.8 m | some | good | some | need fauna | Beech ⁹ |
| brown, drained, moist | 0-0.6 m | no | good | much | pioneer | Ash ¹⁰ |

Table 4.6: *MANAGEMENT PROPERTIES, PART 4; Overview of some of the properties of trees under management and effects of different factors on different tree species. Wood quality is related to the change in quality caused by increased growth rate. The rejuvenation properties are classed as emergent if the tree will emerge as the secondary tree taking over the stand of a pioneer. Weedy implies that it multiplies profusely. Weak implies that it competes weakly with aggressive pioneers. The root penetration of the ground is always limited by the groundwater table, which only alder can penetrate.*

6. harvest
7. recreational activities
8. preservation of biodiversity
9. compliance with boundary conditions such as no leaching of aluminium, nitrogen or heavy metals to downstream recipients
10. environmental degradation

Environmental degradation can be divided into short term cost items and items for the long term perspective. The environmental degradation costs accumulate and develop into long term liabilities in the balance sheet. The cost items of environmental degradation are:

1. Excessive leaching of base cations from the soil (the cause may be chemical erosion by acid rain, physical erosion, excessive uptake by overdosages of nitrogen. Replacement cost at 1,500-3,000 kr/ton)
2. Nitrogen leaching to surface waters (unclear how the cost is defined, loss of nutrient ? damage to recipient ?)
3. Aluminium leaching to surface waters (unclear how the cost should be estimated, liability for damage to habitat ? Restoration costs ?)
4. Release of greenhouse gases (unclear how the cost should be estimated)

| Trees species | Fuel wood Price in m ³ | Pulp wood spruce equivalents | Timber m ⁻³ |
|---------------|--------------------------------------|---------------------------------|---------------------------|
| Norway spruce | 0.6 | 1 | 1.6 |
| Scots pine A | 0.6 | 0.8 | 2 |
| Scots pine B | 0.6 | 0.8 | 1.2 |
| Birch A | 0.6 | 1 | 2 |
| Birch B | 0.6 | 1 | 1.6 |
| Beech Veneer | 0.6 | 1.1 | 6 |
| Beech A | 0.6 | 1.1 | 4 |
| Beech B | 0.6 | 1.1 | 2 |
| Oak Veneer | - | - | 13 |
| Oak A | 0.6 | - | 6 |
| Oak B | 0.6 | - | 3.6 |
| Oak C | 0.6 | - | 2 |

Table 4.7: *MANAGEMENT PROPERTIES, PART 5: Approximative relative price structure per cubic meter for the North European markets, normalized to the Norway spruce pulp wood price (SPE; Spruce Pulpwood Equivalents) for different tree species was used in this study. The market price in year 2000 was used for the scaling. A, B and C represents sawmill quality classes, from A is best and C is the poorest. The above table can be used to convert production to a wood equivalent unit in long term calculus of profitability. The assumptions in this are that the price ratio between different species stay constant and that one cubic meter of harvest is equally interesting every year during the rotation period. This avoids the uncontrollable escalating uncertainty of using an arbitrarily chosen interest rate of 100 years.*

5. Release of heavy metals and environmentally persistent pollutants (unclear how the cost should be estimated)
6. Loss of sustainable production potential because of acid rain effects (loss of potential income ?)

It remains to define the costing methods for these items. The cost of base cation leaching is the a term where a clear method is proposed. In costs, we need to account for any occasional decrease in soil nutrient storage (base cations; Ca, Mg, K as well as N and P). This is a very substantial item and cannot be overlooked in good book-keeping (Sverdrup and Rosen 1998, Warfvinge et al., 1992). Soil nutrient overuse, in excess of what weathering and deposition can supply, will be taken from the internal storage. The cost is estimated from the replacement cost. There are commercial companies that can replace Ca, Mg, K, P and N in the soil at a price. Some will also replace certain micro-nutrients like boron, copper, cobalt, molybdenum if necessary. Thus, for production at a level above the sustainable, there will be a decrease in internal resources which we convert to wood production, the increased growth will be able to do it. It is important to realize that we can easily convert soil nutrients from the storage in the soil to the storage in standing wood, but that this may under certain circumstances cause a decrease in the soil storage as a consequence of mass balancing. Thus, we can transfer value from storage in the soil to standing tree stock. This implies that we transfer a reservoir from

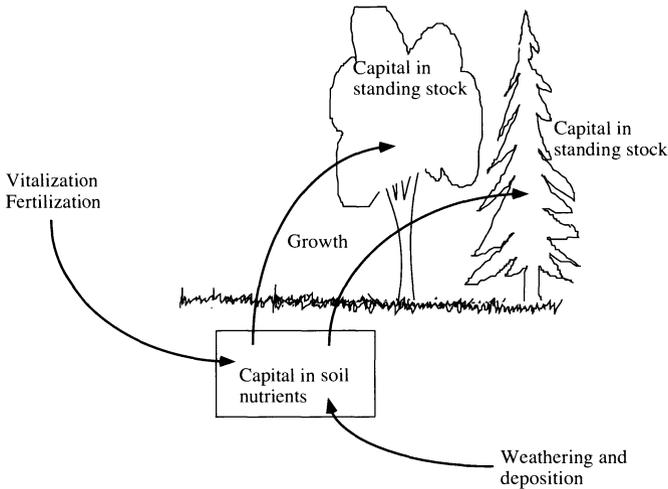


Figure 4.4: *The forest management can move invested capital between the soil and the standing stock. Keeping capital in standing stock of trees have high availability, but also high risk (Pests, pathogens, wind damage, environmental pollution). Storing capital in the soil has lower risk, but lower availability. A minimum amount of nutrient capital is required.*

non-harvestable to harvestable (Fig. 4.4). The price for the nutrient pool is estimated at a replacement price of 3,000 SEK per hectare, including base cations, phosphorus and nitrogen, in Tab. 4.8 we have developed the example further and compared it to land market value. In many cases the nutrient replacement price will be higher, considering the topography and road availability.

How the reestablishment of rejuvenation is discounted after harvest, is also of importance. At present, the rejuvenation cost is in reality, immediately written off in its entirety against the harvest profits, and this is not projected forward to the future harvest. Thus, the cost of rejuvenation belongs to the completion of the harvest of the past crop, being an integral part of this. The forest law is at this point inaccurate, and at this specific point in need of revision. Natural rejuvenation is used by many forest owners, but this is also not entirely without cost. Depending on nutrient return need, additions are best added when the harvest is done, and additional treatment of the land may be necessary for preventing root rot. In soils the stock value of nutrients can be illustrated with Tab. 4.8, based on the replacement cost for the nutrients, assuming we pay somebody to put them back. The costs are taken from the experiences made in the different Swedish lake, stream, soil and forest liming programmes. It is estimated that Sweden has used more than SEK 2,000 million SEK on liming of aquatic ecosystems and some adjacent lands. In the actual projects, liming costs ranged from 2,000 SEK ha^{-1} for vitalization and light replenishing of soils to more than 15,000 SEK ha^{-1} for restoration of damaged and severely depleted soils. The market value for a forestry property is in the order of magnitude of 15,000-35,000 SEK per hectare, and the restoration cost on a depleted soil may cost from 10 to more than 100 % of the land market value. This can be compared to the nutrient values listed in Tab. 4.8 (Southern Sweden market price, February-April 2000). Forgetting to check the nutrient status of the soil before forest land is bought might prove very expensive !

Establishing a new type of balance

The balance sheet for the forest property will also have some new items in additions to the old. In addition to capital assets, real estate value, standing stock of trees that can be harvested, we would need to consider to add as fixed assets of the property the following items:

1. Soil stock of nutrients specified as at the market price of replacement on site
 - (a) Soil stock of nitrogen
 - (b) Soil stock of phosphorus
 - (c) Soil stock of calcium
 - (d) Soil stock of potassium
 - (e) Soil stock of magnesium
2. Hunting and fishing rights
3. Permanent cultural assets on the property
4. Recreational and public access value

The inclusion of hunting and fishing rights is already being practised by many, and is trivial. The inclusion of fixed cultural assets is far more difficult to do and needs development. Possible ways forward would be a discounting of future cashflow from official subsidy programs promoting cultural preservation. Lack of such programs would turn the cultural assets to a balance sheet liability. The same apply to recreational values and public access. Ecological values are also an asset, but the evaluation of it remains difficult, as well as if it is to be seen as an asset or a liability or both.

1. Soil stock of carbon above the long term retainable
2. Soil stock of nitrogen above the long term retainable
3. Soil stock of heavy metals above the long term retainable
4. Residual obligatory restorations (environmental, cultural, recreational, biological) on the property

The residual restorations may be liabilities created by restoring damaged cultural assets, water and stream restorations, restorations of lost biodiversity required by official regulations, restoration of public access or restoration of aesthetic values. Thus, it will be possible to accrue environmental debts over time by careless management, and this need to show on the balance sheets.

4.7.4 The problem of prices and interest rates

A considerable problem arise when the income and costs are modelled over longer time periods. A typical rotation period can last 100 years, and costs and income can occur throughout this time. If care is not taken, the effort may very fast deteriorate to a comparison of exponential tables, yielding no information on profitability. We suggest that this should be eliminated from the calculations to decrease the unreasonable uncertainties introduced by the guesses contained into future price and interest rate projections.

We propose to convert the costs and prices immediately to Norway spruce pulp wood equivalents. This assumes that the harvest of one cubic meter of wood is equally interesting during any year in the whole period. As we have discussed earlier, this allow us to limit the actual interest rate for alternative investments with the sustainability constraints of the system, and thus let sustainability capacity affect the interest rate and the profitability reference frame of

| Nutrient storage loss as a fraction of total capacity | -5% | -15% | -30% | -100% | Land value |
|---|--|---------|---------|----------|----------------------|
| Soil type | Replacement cost, SEK ha ⁻¹ | | | | SEK ha ⁻¹ |
| Organic soils | -4,000 | -12,000 | -24,000 | -100,000 | 15,000 |
| Sandy, mineral soils | -600 | -2,000 | -4,000 | -12,000 | 25,000 |
| Podsols soils | -2,000 | -6,000 | 12,000 | -48,000 | 30,000 |
| Rich soils | -5,000 | -18,000 | -36,000 | -120,000 | 35,000 |
| Agricultural and clay soils | -5,000 | -18,000 | -36,000 | -120,000 | 35,000 |

Table 4.8: *Approximate value for decreasing the nutrient storage in the soil, based on base saturation and soil type. The costs were calculated from replacement value. If the base saturation or total nutrient store should decrease with 5% or 15%-points, then the cost for putting it back in the soil is indicated. A very approximate property value has been included for comparison, just to show the size of the capital hidden in the nutrient pool. The price for the nutrient pool is estimated at a replacement price of 3,000 SEK per hectare, including base cations, phosphorus and nitrogen. 100 SEK=12 US Dollar. A typical weathering rate in a Norway spruce plantation (0.3 keq ha⁻¹yr⁻¹ has a vitalization value of 30-60 SEK ha⁻¹yr⁻¹, a considerable amount over a whole rotation of 70-120 years.*

the system. An endlessly high profit may seem short term desirable, but it goes without saying that it is a very unrealistic dream, and because the earth is limited, alternatives on it will also be limited. The calculus transfers this wood volume forward or backward, before conversion with today's market price is performed. The only necessary assumptions in this approach is that the price ratios between different tree species stay constant, and that the prices for labour or materials stay constant in pulp wood equivalents. These assumptions are far less uncertain assumptions than the uncertainties in those embedded in exponentially escalating interest rate and price projections (Bossel 1998).

4.7.5 Risks

Sustainability requires that the system is robust. In this case, the forestry system must be robust in order to remain sustainable also under extreme or changed external forcing. Extreme weather, or indeed a changed climate, are examples of external forcing that could result in states that are difficult and/or costly to recover from. Hence, such forcing could endanger the sustainability of the system. A robust system, furthermore, allows for freedom of choice concerning management actions to be taken on the system. From a management perspective, events such as those caused by extreme weather are uncertain. In any type of forest management, uncertain events with negative outcome, i.e. hazards, and uncertain events with positive outcome, need to be considered. Handling the uncertainties associated with negative outcome is termed risk management (in economic terminology also uncertain events with positive outcome are included) and includes some kind of risk assessment in which the probability of occurrence of the hazards is estimated. In a formal assessment of risk, the exposure (frequency and magnitude of external forcing) and the response are considered (Bartell, 1996). Such quantitative risk assessments allows informed decisions to be taken without undue influence of the short-term history. One task for SUFOR is to provide decision makers with support in relation to handling risks. According to fundamental decision theory, the decision maker's willingness to take risks affects the decisions. We assume the forest managers are unwilling to take risks, i.e. they are risk averse. This assumption is supported by Lönnstedt and Svensson (2000), at least when larger monetary values are at stake.

Because the decision maker has only partial information on the states of nature, rather than estimating a single probability for the outcome associated with a specific alternative, the decision maker needs to estimate a probability distribution for each available alternative. According to the Gärdenfors and Sahlin (1982) decision theory, a risk averse decision maker would decide on the alternative with the largest minimal expected utility. For complex systems, such as the forestry system, a reliable estimate of the probability distribution for a specific alternative is difficult to make in one's mind. Because of this, we use computer models in order to make

more reliable estimates of the probability distributions than can be made without the models (Gärdenfors, 1982), thus aiming at reducing the uncertainty in the decision situation and to explore the possibilities for reducing the risk (probability) of hazards occurring. Extending risk not only to multiple choices and in time but also distributed in the landscape, become a very complex process requiring sophisticated models.



Figure 4.5: *Cultural remains are protected in Swedish law, and forest management must not interfere or cause damage. Cultural remains are an asset of the property, and their conservation is a societal responsibility, partly transferred as obligation to the landowner under certain circumstances. It becomes a liability towards the authorities if damaged, an asset if it can be subject to cultural conservation support money. Photo: Peter Schlyter.*

4.7.6 Future development

A user-friendly economic module is envisioned as part of the SUFOR computer tool package. The package is intended to perform certain tasks:

- Estimate the sustainable production from simple forest management planning information and a few key soil properties (ForSAFE will be the biogeochemical module)
- An ethical budget; Estimate how well societal responsibilities and ethical obligations have been met and what the real costs for doing so was. A kind of counting promises held and good deeds done.
- A green budget; Estimate the profitability in wood equivalents
- A green balance; Estimate the stock of capital in wood equivalents
- Projected total budget and balance converted to present monetary value

The intention is that the computer programme will be designed to easily fit on a CD and is to be used on home computers. This part will be realized under the second part of the SUFOR programme. The package will be available free of cost for any forest owner, probably download from the SUFOR homepage on the internet.

4.8 Concluding remarks

A complete rethinking of the presently prevailing forest management philosophy is needed. Considering the past history of Sweden and the change of forest management paradigm every 50 years, it appears as the time for a change has come again. With the changes ahead, this offers a large possibility for renewal and innovation. We predict that the new information technology will also slowly enter forestry with instant information access, including sophisticated advice, pricing and sales. The SUFOR programme alone cannot do this, but it can have an important role for initiating and promoting the necessary learning process. A holistic perspective towards system dynamics must be adopted, and we need to work at different scales. This calls for a smooth cooperation and due respect for the multitude of interests and stakeholders involved.



Figure 4.6: *In many countries, large clear-cuts cause outcry in the public. The forests on the American Pacific coast have a history of careless logging in large cuts on slopes with following loss of soil and deterioration of the forest ecosystem. This is no longer tolerated by the general public and are examples of how a certain short term economically profitable practice may become socially unsustainable, and eventually subject to law regulations (M'Gonigle and Parfit 1994). Photo: M. Svensson.*

Chapter 5

Integrated modelling

Patrik Wallman, Harald Sverdrup, Mats G. E. Svensson and Mattias Alveteg

5.1 On the principle of modelling in general

Models are highly important in research, not because they produce results in their own right, but because they allow complex and non-linear systems to be investigated and data from such systems to be interpreted. With models, the interaction of several simultaneous processes in a single experiment can be studied. Basically models serve one or both of the two following aims;

- Testing the overall understanding one has of a system, on the basis of a mathematical representation of its subsystems and the proposed coupling of its subsystems.
- Predicting the future, so as to explain how and why things have worked in the past.

A model represents an integration of the modeler's knowledge and understanding of a system. In some cases a researcher bases a model on equations and parametrization, testing the understanding of a system empirically. In such modelling there are no "maybe's", parameters being assigned quantitative values according to precise rules. At the same time, as many problems related to natural systems are complex, non-linear and multi-dimensional, non-linear models are being required. The solutions in such cases are often beyond the reach of simple empirical approaches, linear regressions, descriptive efforts or paper-and-pencil-only approaches. Examples of complex problems of this sort include problems of the effects of eutrophication on population dynamics and phosphorous cycling, the acidification of soils and its effects on forest growth, nutrient cycling in vegetation, heavy metal mobilisation and global climatic change.

A model can be a consequence or an interpretation of a set of observations or simply experience. A good model has one of the following characteristics:

- It is transparent. Its being possible to inspect and understand the rules and principles the model is using.
- It can be tested, its using inputs that can be defined and determined, and outputs that can be observed.

A model can represent the mental understanding one has of a mechanism, system, pattern or principle, and can be substantiated at the same time be expressed in the form of an equation or a set of equations or rules. If the principles and rules are many, it is often best to let a computer program keep track of all the connections and numbers involved. The goodness or badness of a model does not necessarily have anything to do with the adequacy of the principles it contains. If a model is good, one can verify or falsify its performance. In modelling, the total complexity of the system is found partly in the assumptions and partly in the model itself. For every question there is an optimum, and great care should be exercised in evaluating it. Failing

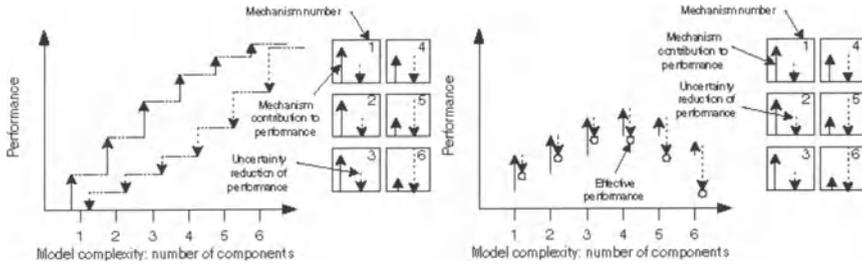


Figure 5.1: That complex models are better than simple models is a general misunderstanding. Often the simple additive build-up of performance is envisioned, forgetting the accumulation of uncertainty as seen in the left figure. Performance build-up in a model is shadowed by the decrease in performance by uncertainty accumulation. It becomes apparent that model performance is best when the difference between the accumulated performance and the accumulated uncertainty is at maximum, as showed in the right figure. Thus, a simpler model may perform significantly better than a complex model with many components.

to do so can result in the loss of control over uncertainties. It is important to realise that one cannot get rid of the complexity of a system, but can only decide whether it should be assigned to the model or to the assumptions.

- A simple model can make very complex assumptions. These should be stated explicitly. Although simplifications may make a model highly complex, use of such a model has the advantage of requiring relatively little input data.
- A complex model can make very simple assumptions and accordingly require a large amount of input data. Adding further complexity to the model may remove assumptions and take account of more feedbacks but place higher demands on the input data.

All models must fulfill certain minimum requirements. They need to be able to describe events at individual sites on the basis of real data. If a model cannot describe single sites and their past history, it has no credibility for future predictions. If a model is capable of reproducing the past history successfully, predictions for the future have better credibility. Similarly, if a model is unable to reconstruct the past in an acceptable way, it lends no credibility to predictions for the future.

The idea that complex models are better than simple ones represents a misunderstanding. Often, a simple additive building up of how the overall process will proceed is envisioned, it is often being forgotten that the uncertainty associated with each of the components accumulates. A model performs best when the difference between the accumulated performance and the accumulated uncertainty is at maximum (Fig. 5.1). As each component in a model is added, performance can at one level be said to improve at the same time as a reduction in the resulting performance occurs due to the uncertainty that is added. When a component brings more uncertainty into the model than an improvement in performance, there is no net gain in performance being added through the component. Thus a simpler model may perform better than a complex model with its many components. Going to the extreme, a very complex model may show no performance at all, being impossible to apply. Thus, the statement "we want a model that represents the full complexity of of the system" may be a good recipe for modelling failure. The terrestrial flora and fauna of a catchment is important for the cycling of nutrients, such as base cations, phosphorus and nitrogen, as well as for the balance of acidity there. Major biological processes to consider in this context are the growth and harvesting of plants, the decomposition of organic matter, nitrification, immobilization and denitrification.

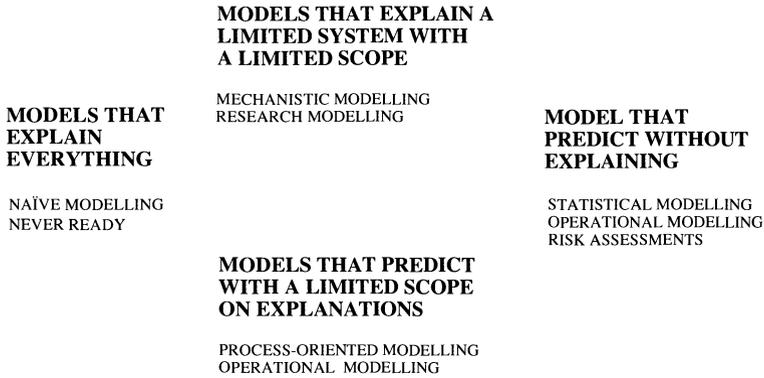


Figure 5.2: *Survey of different types of models. For the SUFOR programme, process-oriented models of limited scope are preferred due to their greater versatility and ease of adaptation to field conditions. Naive modelling should be avoided at all cost.*

These processes all have strong feedbacks with the soil chemistry. For a specialist in a certain field, it may be easy to succumb to the illusion of the process one is currently studying being the only significant process involved. This, of course, is rarely the case, most natural processes stand in close communication with other processes. It can be assumed, as a rule, that natural processes affect each other mutually in an appreciable way, and that other processes can thus not be ignored a priori. Accordingly, a system perspective is needed for progress to be made. In trans-disciplinary natural sciences such as biogeochemistry and ecology, models for different compartments and systems need to be coupled, so as to take account to all significant interactions and feedbacks of the system.

Several different types of models are available, based on different philosophies and levels of ambition. The general level and type of models currently employed has evolved historically in a series of different phases. Levenspiel (1972), a pioneer in chemical engineering, described the development of process modelling in engineering over time in terms of three stages:

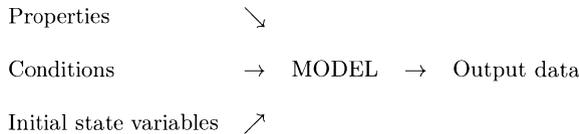
| Stage 1 | Stage 2 | Stage 3 |
|-------------------------|--|---|
| Qualitative description | Direct quantitative description in terms of observable | Differential rate based on underlying physics and processing conditions |
| | → | → |

The classical botany of Carl Linné (1707-1778) with its description of plant types and their geographical distribution is typical for Levenspiel's stage 1. Such a model predicts what kind of plant one can expect to encounter in a certain part of the country, but says nothing of why it is there. Such a model has virtually no predictive power over extended periods of time. It records things as they are at present. Models corresponding to Levenspiel's stage 2, in turn, have a case-by-case predictive power and need to be recalibrated by use of new data each time the initial and the boundary conditions change. Plotting pH over time in an acidified lake is an example of Levenspiel's level 2 model. Such a pH model is only valid for that particular lake, and cannot be transferred to other lakes in which the ecological conditions differ. Stage 3, finally, is the differential approach taken in physics, for example, by Walker *et al.* (1923),

more generally in physics by Einstein (1905) and in chemical reaction engineering by Hougen and Watson (1947). It describes how the change that occurs at every point in time is related to the state of the system at that time. Stage 3 thus implies there to be a mechanism of change, one which depends on the state variables for the system and is valid generally. The state of the system is characterized by the state of the ecosystem in terms of biological order, demography and ecological adaption, as well as the chemical state of the water and the soil. The active mechanisms are dependent on the properties of the reacting medium and not on the location of the process in time and space as such. A stage 3 model is generally valid and applicable when it can be parameterized properly and its coefficients can be estimated, but it is in differential form, requiring mathematical manipulation to be used. It is generally accepted that models take data in order to generate new data:

Input data → MODEL → Output data

In reality, models use three types of input data: state variables that describe the state of the system; conditions, which are influences and conditions imposed on the system from the outside, and finally properties, or the inherent characteristics of the system characteristics that do not change within the time horizon. Models are either based on statistical regressions involving state variables and conditions or they are mechanistic models involving the application of properties, conditions and initial state variables to hypothesis-based mechanisms and to structures of mechanisms. Process-oriented or fully mechanistic models require no data in terms of time variant state variables, but involve properties, conditions and initial states.



A model can also consist of a combination of one or more mechanistic models and one or more models based primarily on statistical regressions, an approach particularly effective if there is no feedback between the core model used for general conditions and a given subsystem. An example of this would be combining of an elaborate geochemical and hydrological acidification model with empirical correlations concerned with fish survival under the conditions involved. Since there is no feedback of the survival of fish into the geochemistry of the soil, the correctness of the latter is unaffected by the survival of the fish, the correlation involved being fully adequate.

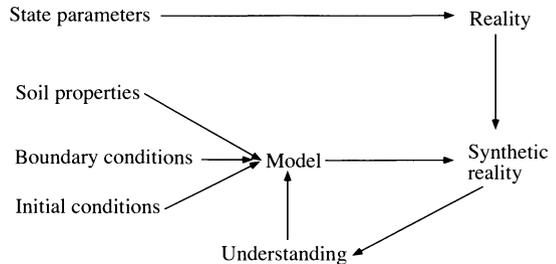


Figure 5.3: A mechanistic model stratifies the "data" into different categories. The model uses properties, initial state and boundary conditions to act on principles and on mechanisms to predict the state variables. The state variables are saved from the data for later validation of the models performance. This emulates the learning cycle shown earlier.

In modelling, there is a strong "cult of success" that implies that any output calculated should pass through all the observation points, its otherwise being concluded that "The model is wrong" or "It is obvious that the model doesn't work". On the other hand, if a model is calibrated by adjusting several parameters, critics may soon remark "You can probably make the model fit anything". Although calibration

is necessary, it should be used with great caution, so as to avoid misinterpretations. If a process model is calibrated on more than one or two processes, it becomes much more likely that errors in model formulation will be concealed. This can result in the calculations looking correct but it is not being possible to distinguish between artefacts of calibration and the correctness principles the model contains, a good fit being no help then in understanding the system. Thus, it is always important to reflect, asking such questions as "What was the objective of the model's being applied in the first place?". As a general rule, understanding of processes of a system and their functionality increases more as a result of modelling failure of its fit being good. Thus far, most dynamic soil chemistry models have been calibrated either on two parameters, namely the weathering rate and the initial base saturation (de Vries and Kros 1989) or on a large number of different parameters such as weathering, initial base saturation, selectivity coefficients, Gibbsite coefficients, uptake and organic complexation. (Christophersen, Seip and Wright 1982, Cosby, Wright, Hornberger and Galloway 1985, Gherini, Mok, Hudson, Davis, Chen and Goldstein 1985). At the same time, interpretation of the basic principles behind soil processes become pointless if one calibrates a larger number of parameters than of output variables. The credibility and robustness of a calibration procedure is greatly improved if historical soil chemistry data for a site is available. In most cases in which soil chemistry models have been used, with the exception of various sites in Rothamsted (Goulding, Johnston and Poulton 1987, Goulding and Johnston 1989) and a few Swedish sites (Falkengren-Grerup and Eriksson 1990, Warfvinge, Falkengren-Grerup and Sverdrup 1993, Sverdrup, Warfvinge and Nihlgård 1994), no historical values for soil chemistry, except for those at only one point in time (the present) are available for model validation. Data on short-term soil chemical responses are often available, however. Considering an acidification model here illustrates the calibration procedures involved. Adjusting the weathering rate in the model changes the slope of the line for the ANC for the soil solution, as well as influencing the final ANC level. Several soil chemistry models employ the base saturation and the weathering rate for calibration. In the soil chemistry model used in SUFOR, only the initial base saturation is employed, its determining the initial starting level for calculations. The higher the initial base saturation level is, the greater the number of base cations is that needs be leached from their ion exchange positions, this influencing the time and the rate of change required for reaching the final state, the initial state being used to shift the curve either up or down. If the calibration of base saturation and the weathering rate are combined, it is always possible to calibrate the model in such a way that the curve goes through the current observation point, since soil chemistry models normally include charge balances. If further calibration on additional processes such as uptake, decomposition of organic matter and ion exchange is performed, the model is able to fit absolutely anything. If one is certain that processes other than the processes used for calibration are correct, the model can serve as a measuring instrument for determining the coefficients of the uncertain process. If calibration is based on more than three processes simultaneously, there is likely to be a large number of different ways of modelling the data set so as to make it fit the observations, making any conclusion regarding the catchment processes or whatever involved run the risk of being meaningless. The model is no longer a mechanistic model then, but becomes simply a non-linear regression. Thus, before a model is calibrated, care should be taken to consider what kind of information may be lost in the calibration procedure. For testing process formulations, calibration involving more than one degree of freedom should generally be avoided completely, since calibrated runs that ignore certain important processes can be made to look like runs in which there is full process representation. In summary, development of a model never starts with collecting all of the data. It starts with defining the question to be answered or the phenomenon to be explained, as shown in Fig. 1.3. Experience suggests the approach of starting with the collection of data to be an excellent diagnostic trait for spotting unsuccessful efforts to develop a model.

5.2 Modelling the forest system

5.2.1 Introduction

The forest ecosystem stretches from the topmost layers of the atmosphere, where ozone and pollutants can be found, to the lowest layers of soil affected by the roots. It is thus very complex and diversified, many factors influencing the system in ways that are difficult to predict. It is very important, therefore to define the system and its boundaries. The aim of a model determines the complexity, and the temporal and spatial scales that are appropriate. On a regional scale, it is not meaningful usually to endeavour to model every physiological detail. On the other hand, when modelling on a sub-individual level, using time steps that are too long can lead to important information being missing. One way of dealing with this problem is to divide a model into several sub-models that can operate individually on different time scales.

Three main things determine the growth of a tree or a population of trees in the absence of disturbances and pollution: the availability of light, of water and of nutrients. How much a tree grows depends on its ability to assimilate carbon through the canopy, and in what manner it grows depends on the carbon allocation within the tree. The carbon assimilation is a function of the temperature, the nutrient content in the leaves and the incident radiation, the processes involved being regulated by the availability of water. These are the major driving variables in most physiologically based forest growth-models. Water availability is important both for the photosynthesis reaction and for the nutrient uptake from the soil. The soil is not an unlimited source of nutrients as most models assume, at least not in a long-term, sustainable perspective. If the consumption of nutrient by the growth and harvesting of trees is faster than the supply of them from the litter decomposition cycle and weathering of minerals in the soil, then the soil will eventually be either intermittently, or in the worst case, steadily and permanently depleted. In southern Sweden, deposition provides large amounts of nitrogen, the forests being more or less fertilised from above, the rest of the nutrients needing to come from the soil. In several cases, this has already led to nutrient depletion of the soil. The soil and weathering are thus very important parts of the forest ecosystem and cannot be neglected in any adequate long-term forest-growth model. Substantial areas of Sweden have been turned over to forestry during the past century. In individual stands, human activities influence species composition, cover, age, and density. At the landscape scale, people alter the kinds of stands that are present and their spatial arrangement. At the regional level, people emit by-products in the air that may either fertilise or damage forest ecosystems. At the global scale, human consumption of fossil fuels has increased the atmospheric carbon dioxide levels and possibly changed the way in which carbon is distributed in the vegetation, the soil, and the atmosphere, which has long-term implications for the global climate. The growing demand for forest products worldwide has stimulated not only the transfer of processed forest products from one country to another, but also the introduction of non-native species and proveniences. Pressure to extract resources in a sustainable way has raised a number of challenging questions, two of which are the following: Is it possible to maintain a high timber production in a sustainable way? What are the limiting factors?

5.2.2 Forest model history

Quantitative models and methods have been used in forestry for a long time, originally in the form of growth and yield tables derived from large amounts of field data. Empirical models using statistical methods have been developed to reproduce field observations in an accurate and robust way. Empirical models play a very important role in forest management today. Process-oriented models have a fairly short history. During the past decade, many process-oriented ecosystem models were constructed, primarily as educational and research tools. Building on earlier work in the 1960s (McArdle, Meyer and Bruce 1961), many simple forest growth models were developed in the 1970s to serve as predictive instruments for forestry management (Kimmins 1997). The empirical regression models are reliable and useful as long as all ecologi-

cal and physical conditions that influence forest growth remain the same as when the calibration data set was collected. This is very unlikely to be the case over time spans of several centuries, however since models of this sort do not deal with dynamic response mechanisms, they are limited as predictive tools for analysing forest ecosystems under changing environmental conditions. The first successful computer simulation of forests was the JABOWA model developed in 1970 by Daniel B. Botkin, James F. Janak and James R. Wallis (Botkin, Janak and Wallis 1970).

Forests are characterized by species that are long-lived, robust, and very often located on soils with limited mineral reserves. Under these conditions a series of adaptive mechanisms has developed, resulting in very interesting performance of these plants with respect to their overall capacity to use their nutrient efficiency for the production of vegetative biomass. The needs of forests for nutrients for the development of biomass are very high, sometimes surprisingly so, although a series of recycling mechanisms in the ecosystem permit optimum use of the available supply. These mechanisms represent the biological and biogeochemical cycles respectively, of the nutrient elements. To really understand these cycles, very long time series need to be studied or use be made of predictive forest models that are adequate. Because of the characteristic longevity of forests, a forester should be familiar with dynamics of stands, knowing how the stand dynamics are influenced by silvicultural choices made. The forester should also be able to anticipate the evolution of a stand with in view of decisions that are made regarding it. Today, problems of this sort are solved by means of simulations based on growth models. Such forest models are used to predict production-yield, succession and long-term variations within forest ecosystems. A major advantage of computer models is that they produce alternative forecasts in the form of "data" that are consistent with respect to the underlying dynamics of the understanding embedded in the model across the full range of alternative forest management schemes examined. Each forecast reflects the initial conditions found in the forest, the rules of change for stand dynamics as defined in the model, and the responses to the interventions made as defined in the model.

The accuracy of a forecast is dependent on the degree to which the initial conditions, the stand dynamics, and the mechanisms of responding to treatment as stipulated in the model reflect the reality of the forest.

5.2.3 The space/time realm of forest ecosystem modelling

The modelling of forest ecosystem is a mixture of biogeochemistry, ecophysiology, and micrometeorology, one that emphasises the circulation, transformation, and accumulation of energy and matter through the medium of living organisms and their activities. Ecosystem analysis is concerned less with diversity of species involved than with the contribution that any complex of species makes to the transfer of water, carbon, nitrogen, energy, and nutrients within the ecosystem. Ecosystem modelling considers not only the flux of energy and matter through the forest, but also transformations that occur within the forest. Forest ecosystems are open systems in the sense that they exchange energy and materials with neighbouring systems, including adjacent forests, aquatic ecosystems, and the atmosphere. The exchange is essential for the flow of energy and matter. A forest ecosystem is never in complete equilibrium. Although we aim at modelling forest ecosystems across multiple time and space scales, we initiate our studies at the forest stand level, where most measurements and most understanding originate (Aber and Melillo 1991). Most scientific understanding of ecosystem processes has been gained from direct field measurements and from experiments on small study plots, usually less than 1 ha in site over periods from a few days to several years. In scaling terms, we refer to this as the stand level. Modelling studies of this sort aim at clarifying the ecological and physiological processes involved and concentrate on the forest and not on the spatial heterogeneity of the surroundings, or the temporal changes that forests have undergone or will undergo in the future. Many of the major concerns in the use of natural resources however, are not tractable by use of the spatial or time scales applicable at the stand level. Forest management and large-scale forestry generally have an impact on larger and more diverse areas, and it must also prepare for future conditions

and ecosystem responses that may result from policies and decisions made at present. Consequently, to make a forest ecosystem model more relevant, it is essential that knowledge of the processes at the stand level can be extrapolated in space and be extrapolated forward in time. Direct studies of large areas or longer time spans are seldom feasible scientifically or economically. Models often represent the only possibility of making such studies. A set of conceptually linked computer simulation models offers a valid alternative if it can represent the mechanisms that couple biogeochemical processes with the atmospheric and hydrological processes which affect the ecophysiological processes in trees.

5.2.4 Model roles

Ecosystems are complex, with many variables and stakeholders. They also interact in non-continuous and non-linear ways, and are thus too complex to be described by a few equations. In the absence of a satisfactory experience-based method for predicting the long-term sustainability of forest yields and its impact on biodiversity, we need to use knowledge-based prediction systems. Given the necessity of making knowledge-based predictions, and problems of discontinuities in spatial, temporal, and complexity scales, there is a need for an alternative prediction system for forestry. Traditionally, predictions of yield have been based on forward projections of historical patterns of growth. This "historical bioassay" approach is very effective if future growing conditions are similar to those in the past, but it lacks the flexibility to make accurate yield predictions for a future that markedly different from the past. There are several cases in which the yield prediction method of dealing with changing future growing conditions has failed. The scientific response to these shortcomings of the historical bioassay approach has been to develop yield prediction tools in the form of process-based simulation models, that provide a good understanding of important ecosystem processes e.g. (Mohren 1987, Bossel and Schäfer 1989, Kimmins, Maily and Seely 1999). There are certain limitations to the use of this type of models as practical predictors of forest growth and yield. Many process-based models include only a subset of the important determinants of forest growth and yield, and have very demanding calibration requirements. This has tended to limit the application of such models in research and education. Hopefully will these limitations will become less prominent in the future as understanding of the processes involved improves. A third approach to growth and yield predictions is the "hybrid-simulation" type of models, in which the historical bioassay and process-based simulation is combined through either simulating or calculating various multipliers such as light, temperature, moisture, or nutrients that are used to modify historical patterns of forest growth, or through simulating the availability of resources (light, nutrients), and the supply of these needed to enable the historical pattern of growth to recur. The pattern obtained is adjusted then accordance with the resource limitations to growth under higher than expected resource conditions, e.g. the FORECAST model, (Kimmins *et al.* 1999).

It is important that dynamic models be applied to many difficult environmental and pollution management issues where they can be useful. Various problems of the acidification of forest soils are coupled with other problems. The most important of these are the problems of nutrient imbalance in forest soils and its impact on forest ecosystems and surface waters, of sustainability, and of the feedbacks from the increase in carbon dioxide to the climate. Nitrogen deposition is continuing, whereas the sulphur deposition is presently decreasing. Important couplings have not yet been made. The understanding of the systems here and of the responses involved which an integrated systems analysis can provide will be significantly different from the answers provided by analysing each problem alone, although we are already aware of many of the effects that can be shown in this way. Changes in precipitation pattern temperature pattern and the biodiversity are likely to decrease dramatically.

5.2.5 Scaling

One of the major compromises in model scaling we are required to accept is the need hold details to a minimum. This has the advantage of reducing the complexity of a model, but it requires that one gain insight into which ecosystem properties are critical and then determine how they can be measured, condensed, and integrated on a progressively larger scale. Problems of pattern and of scale are central problems in ecology (Levin 1992). Moreover, as Levin (1992) emphasised, the scale at which a pattern is observed is often much larger than the scale at which the process is studied. There are two components to the problem of scaling: identifying the scales that are important and producing an algorithm for relating processes across scales. There are three dimensions along which a scale can be defined: organisation, time and space, all of them having two properties: extent and resolution. A reasonable world view envisions a hierarchical system in which material causation works upwards and set-point constraints works downwards. This view has led to two different approaches to model construction. Bottom-up models attempt to predict high-level phenomena on the basis of low-level processes. Such models are deterministic, mechanistic, and process-based, explaining higher-level system performance as being the outcome of systems on lesser spatial and time scales. Such explanations can be error-prone due to error propagation of knowledge beyond the scales on which it was acquired. Top-down models attempt to describe system behaviour as being the result of relations between system variables and external driving variables (Jarvis 1993). Temporal scaling is important as well, since many processes that take place are temporal-scale-specific, e.g. rapid processes not being noticeable at longer time perspectives, even if the result of a process is obvious. Slow processes, in contrast, are not detectable if the time perspective is short, and may thus not be obvious at all. Accordingly, the choice of temporal scale is dependent on which processes one focuses upon. In the present research project, the ultimate focus is on issues of sustainability, which we have defined as being processes taking place over five forest generation cycles, i.e. around 300 years. Not all processes are measurable at this time perspective, which is still very short for certain critical processes such as weathering. As one increases the temporal extent of analysis, the spatial scale also increases; events that occur uniquely on a small scale spatially occur more quickly than those that occur on scales of larger size. In addition, biological properties covary with spatial and/or temporal scales although the outcome is complex, such as scaling of plant photosynthesis on a scale ranging from biochemical reactions to global primary production. The problem of scaling across space from an existing low-level model can be addressed by four different approaches (King 1991, Hinckley, Sprugel, Brooks, Brown, Martin, Roberts and Schaap 1998): Lumping is probably the simplest and most common approach to changes in scale. It involves retaining the original mathematical model, but selecting new parameter values applicable to the larger scale. An example of this is the "large leaf" approach in which an upscaling is made from a single-leaf-based physiological model of photosynthesis to primary production of the forest canopy. This is the forest modelling approach we have chosen for the upscaling of primary production elements. Direct extrapolation is another approach, the model's inherent spatial unit being replicated a sufficient number of times to provide the larger spatial scale appropriate information and material flow between units. This is the approach we have used in the soil weathering models. Extrapolating by expected value is a further approach that scales local output to a wider region by multiplying the area of the larger region by the expected local output. One problem with this approach is that of defining which of the local outputs to use and how to combine them to form an aggregated variable. Finally, explicit integration is an analytical solution that requires mathematical integration of the local function over two- or three-dimensional space, an approach we used partly in some the soil model approaches. Most of the models presented above have been used for scaling from a single forest stand to the watershed level by use of direct extrapolation method of replicating the area modelled over larger areas. Major efforts have been made in recent years to model spatially explicit landscapes involving variable soil, hydrology, plant communities, and human impacts. These models are complex, since they integrate so many non-linear mechanisms and processes.

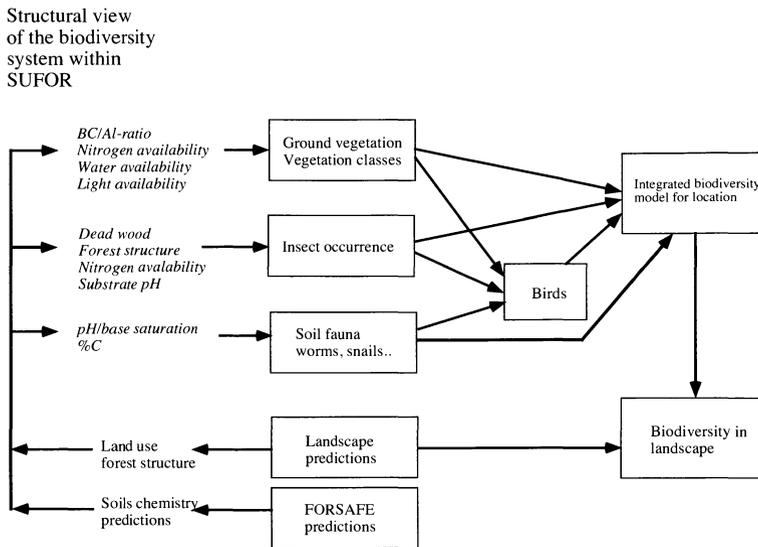


Figure 5.4: A conceptual diagram of a biodiversity system as it could be handled by a model.

They are collectively known as process-based, as opposed to individual-based. These models usually solve the scaling problem by direct extrapolation, moving from small grid cells to larger regions.

The area our models need to be upscaled to is the southern Swedish region, which has a size of approximately 200,000 km². Flexibility between upscaling and downscaling of the input data and of the model employed is also needed for two major aims of the SUFOR project: supplying individual forest owners with a forestry-planning tool and improving methods of regional forestry planning. These aims are guidelines when selecting an appropriate grid size. Climate models usually use a grid size of 100 km x 100 km, or 250 km x 250 km. The models SAFE and PROFILE have been used in grid sizes of 100 m² up to 150 km x 150 km.

5.2.6 Different model approaches

Forest models can be divided into three groups: physiological, architectural and mass balance based. Models within the three groups are structured in similar ways. A model can also be categorised tree-, stand- or landscape-level oriented. A physiological model has plant-physiological relations as its main determining structures in calculations; the model's relations describe in a biologically correct way what is happening in the plant or plants. Architectural models describe plants or ecosystems by use of a hierarchy of geometrical models. Trees are described as being polygons with specific growth characteristics. A mathematical model is thought to predict tree or forest development by use of empirical mathematical equations. A model can also be categorised on the basis of resolution level. Stand-level-based models are constructed on the basis of information at a stand level, such as tree density or total basal area. Individual-based models describe trees as being unique entities, that become established, grow and die. Landscape and ecosystem models take not only trees but also other elements of vegetation into account.

Succession models or gap models are usually employed for describing ecological patterns and processes of forest dynamics over long periods of time (Liu and Ashton 1995). Gap models are used to describe the physiological and ecological behaviour of trees over time. The lifespan of

Structural view of the stress model within SUFOR

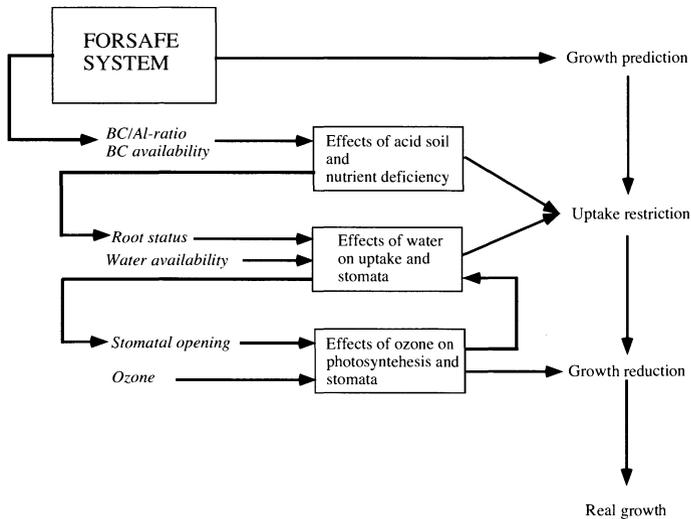


Figure 5.5: A conceptual simplified diagram of the stress model structure.

an individual tree can be divided into three phases: establishment, growth and decline to death. Tree regeneration can occur through replantation or natural regeneration. In gap models, the conventional way to introduce new individuals is from an external seed source (Shugart and Noble 1981). The success of the seeds that are released depends on the site conditions and such species-specific factors as seed phenology, seed longevity, dormancy periods, seed dispersal mechanisms, the amount of viable seed produced and shade tolerance. A general assumption for gap models is that the climatic conditions are within the framework of the degree-day concept (Shugart and Noble 1981). The species selected to initiate succession in the model and its juvenile numbers are assumed to be chosen stochastically from a specified external source (Liu and Ashton 1995). Tree growth is expressed as increase in diameter, height, basal area, root weight and crown size. Most models are designed for simulating diameter growth, height and basal area can be derived from species-specific functions (Hilt and Teck 1988). Growth rates are strongly influenced by site conditions and by interactions between individual trees. The initial gap model of Botkin *et al.* (1972) assumes the diameter of a tree to increase at a maximum rate under optimal conditions. The increase in diameter is influenced by such factors as life history characteristics, maximum age, maximum diameter, maximum height and the fact that the tree growth is influenced by a tree's abiotic and biotic environment. Mortality can be classified as being regular or irregular. Regular mortality is caused by suppression or competition for limited resources. Irregular mortality occurs due to density independent forces such as insect and pathogen attack and catastrophic factors such as hurricanes, floods or fires (Liu and Ashton 1995). Irregular mortality is treated in models as being a stochastic process. Gap models simulate the mortality caused by suppression, random factors, harvesting and aging (Botkin, Janak and Wallis 1972). In gap models, the aging mortality for an individual over a given period of time is a function of the maximum age of the species. Sudden catastrophic event-mortality is integrated into many gap models.

In contrast to gap models, growth-yield models often simulate forest production in a shorter

time perspective, regeneration by plantation. They usually describe growth rate as being a regression function of such variables as site index, basal area and tree density. The site index, a measure of site quality, is generally expressed as the expected height of the dominant and/or co-dominant trees at a specified age (Clutter, Fortson, Pienaar, Brister and Bailey 1983). The primary assumption made is that the factors influencing growth can be integrated into single site index (Meldahl, Bolton and Eriksson 1988) expressing is the outcome of interactions between different environmental factors such as nutrient and water availability rather than the environmental factors themselves. It is used to evaluate the maximum growth rate or the growth potential. The maximum growth is modified by factors such as tree size and density. Not only the site index but also the competition indices influence growth rate appreciably. Many growth-yield models use competition indices to measure the competitive impact of adjacent trees, incorporating these competition indices into predictive models for estimating individual tree growth. Competition indices are based on such factors as: diameter at breast height, stem position or total basal area. The growth functions vary with respect both to the type of function involved and with respect to the variables that are included. Growth-yield models do not simulate death due to aging, since there is no need to do so for timber trees harvested prior to senescence. Lin (1974) used a regular mortality rate that varied with the growth space index. Another way of modelling mortality is to use a logistic mortality function. There are two ways of harvesting, selective harvesting and clear-cutting. Selective harvesting involves cutting trees greater in diameter than some diameter decided or on the basis of some other criterion. Clear-cutting means all trees being harvested. Death caused by insect pests is a major contributor to mortality in managed forests, the use of proportional hazard models being a way of dealing with this.

5.2.7 Overview of selected models

Many models are available, the following not being intended as an exhaustive list or review. Rather, it represents some of the models we have used as a source of inspiration and experiences in our own efforts at model development.

JABOWA

The origin of gap models was a model developed by Botkin *et al.* in 1972 called JABOWA. A model of the JABOWA family and its ancestor FORET consists of one or more spatial cells occupying 0.01-0.1 ha. There are many trees in each cell, their spatial arrangement not being determined. The model consists of sub-models for the growth, mortality and recruitment of each tree and of the available resources. The model is based on current physiological knowledge. It is capable of predicting critical features of natural forests. In the JABOWA model, trees compete for light, soil water and soil nitrogen. In the growth sub-model of JABOWA, individual trees show sigmoid growth under ideal conditions, the final asymptotic size being species-specific. A tree's actual growth rate is obtained by multiplying the tree's ideal growth rate by a series of species-specific growth modifying functions describing the effects of resource availability and climate on growth (Pacala, Canham and Silander 1993). A limiting factor in the model is shade tolerance. A tree is placed in one of three shade tolerance classes and is given a specific light index. In JABOWA, Lambert-Beer's law governs the light availability. Most models of the JABOWA type involve mortality being based on empirical relationships. Resource-dependent mortality is initiated when the growth rate slows down to less than 10% of the growth rate under ideal conditions. The trees in the model do not produce juvenile trees, but rather are drawn from a fixed list in each iteration. Dispersal and abundance-recruitment feedbacks are not included. In some models, the herbivory influences recruitment. JABOWA II and III are two recent versions with improved capabilities.

Treedyn

Treedyn is a physiology-based process model. It consists of ten ordinary non-linear differential equations, one for each state variable. The state variables are leaf biomass, fine-root biomass, wood biomass, fruit biomass, carbon in the forest litter, carbon in the soil organic matter, nitrogen in the forest litter, nitrogen in the soil organic matter and nitrogen in the soil available to the plant. The major processes governing these variables are radiation input, canopy photoproduction, respiration, leaf and fine-root renewal, carbon allocation and increment, litter fall, nitrogen demand and supply, decomposition, and nitrogen mineralisation (Bossel, Krieger, Schäfer and Trost 1991). Daily changes in canopy solar radiation, seasonal maximum noon radiation input, length of the day and temperature are modelled as time functions. These functions are parameterized with regional parameters. The canopy is divided into several leaf layers. The number of layers is related to the leaf area index, which is determined by the net primary photosynthesis of the lowest leaf layer. The full daily photosynthesis and respiration is simulated to obtain the net respiration rate at any given time. For each simulated time of the day, the current radiation over the canopy is determined first. The light received by each leaf layer is found by assuming exponential attenuation, the light-extinguishing coefficient being species-specific. The current photosynthesis of each leaf layer at the current radiation level is then determined from the species-specific net photo-productivity curve. The exponential approximation of the photo-productivity curve is parameterized in terms of maximum photo-productivity at light saturation, the light saturation point, light compensation point and the dark respiration.

The current total canopy production rate is obtained by integrating over the leaf layers and the hours of the day for a full daily cycle. Respiration of the wood, the fine roots, and the fruit, and nighttime respiration of the leaves are considered as drainage of the assimilation. The wood respiration is related to the sapwood fraction, which is determined by geometrical relationships in the tree. Hierarchical assimilation employed in the model, subsequent allocation being assumed. Respiration needs have highest priority, leaf and fine-root replacement the second priority. Above a certain nutrient level that is needed for incidents, fruitification can take place. If the assimilation reserves are sufficiently large growth occurs. Construction allocation is accounted for in these allocations. To avoid excessive nutrient loss through litterfall, a certain withdrawal of these nutrients occurs prior to shedding. Some carbon and nitrogen is also lost by fruitification and wood removal. The model determines current nitrogen availability and compares it with demand. If the full nitrogen demand cannot be met by relocation, uptake and fixation, the growth of the fruit and of the leaves and both the increment and the photoproduction of the canopy are changed accordingly (Bossel *et al.* 1991). In the litter composition process, a fraction of the carbon in litter is transformed into stable humus compounds, the rest departing as carbon dioxide. Nitrogen from the litter is released at a rate proportional to the soil C/N ratio in the humus. The decomposition and mineralisation are temperature- and site-dependent. Only eco-physiological parameters are included in the model. The parameters can be divided into tree-specific, region-specific and scenario parameters. The model has been applied to a Norway spruce stand in central Europe under various management regimes.

FORECAST

FORECAST NAVIGATOR consists of the ecosystem model FORECAST combined with the user-friendly PC-interface Navigator for Windows. The package was developed at the University of British Columbia, Canada by Hamish Kimmins and co-workers (Kimmins and Scoullar 1994, Kimmins *et al.* 1999). FORECAST is based on the former FORECYTE-11. The model is adapted to the ecosystem level, its being a hybrid between a process-oriented and a higher-level simulation model. It has been applied to Douglas fir stands in British Columbia, Canada. The FORECAST forest ecosystem management simulation model combines a traditional bioassay modelling approach with process-based simulation modelling to provide a method for projecting

future forest biomass yield and a variety of other ecosystem variables and social values under a range of management conditions. The major driving function in the model is shade-corrected foliage nitrogen efficiency, which in turn is limited by the specific site quality. Prior growth records are used to determine the site quality. Many of the representations of soil processes in FORECAST are simplistic. The model is based on four basic assumptions, which if not met, reduce the applicability of the results. The first assumption is that a historical bioassay provides the best estimate available of the growth potential of the species being simulated. The next assumption is that the best method to simulate the consequences for growth allocation and biomass turnover of environmental changes over time is to use data on patterns of allocation and turnover from environments with a range of values applicable to the environmental variables of interest. The third assumption is that the initial conditions of the plant community and the soil can be accurately defined. The last assumption is that the method used to estimate certain process rates indirectly from input data produces acceptable estimates of these.

The Pinogram model

PINoGRAM stands for PINE Growth Area Model. It predicts the proportions of each individual tree within an even-aged stand at a continuously changing degree of interaction with neighbours at any given age. The tree coordinates, tree height, crown length and crown width in several directions are calculated and are drawn. The model also computes classical yield data. It does not use data on either tree architecture or forest stand level. The attainable height of a tree is determined randomly within a range entered by the user and is considered as being the result of genetic and environmental influences (Leersnijder 1992). Height growth is calculated by use of an equation that describes the mean height of Scots pine in the Netherlands, which is derived from statistical measurements of growth. The Pinogram model acts on a level between that of the organism and of the eco-unit, the output being expressed in dimensions compatible with both. The growth factors used on this level are the genotype, the site and the growing space. Each factor consists of several sub-factors and of their interactions. These sub-factors are treated as black boxes.

TREGRO

TREGRO is a forest model at level of the individual tree. It is a deterministic simulation model, which describes the cycle of water, carbon, and nutrients throughout the forest ecosystem (Weinstein, Beloin and Yanai 1991, Tiktak and van Grinsven 1995). The model can be classified as being process-oriented physiological model. Atmospheric deposition is considered an important boundary condition. Throughfall is another of TREGRO's boundary conditions. Soil water transport is numerically integrated by means of a sub-model which includes Richard's equation. In addition, TREGRO includes a module for the simulation of macro-pore water flow. Solute transport is calculated as the product of soil water flux and chemical concentration. Evapotranspiration drives the uptake of water and nutrients and reduces the leaching of water and solutes. The conductance of the forest canopy is explicitly modelled by the Penman-Monteith equation. The stomatal conductance needed as a function of time is derived from the net photosynthesis, the vapour pressure deficit and the foliar water status. The root water uptake flux is simulated by use of the hydraulic conductivity of the soil, the average distance from the bulk soil to the root surface and the hydraulic gradient from the bulk soil to the root surface. The average distance from the bulk soil to the root surface is calculated from the fine-root density. Although the tree growth relations considered are physiologically based, they are often lumped together. The boundary condition for the model is that of carbon availability. The daily canopy photosynthesis is calculated by integrating the carbon assimilation over a number of layers in the canopy, over a number of ages of leaves, and over the course of the day. The photosynthesis rate is calculated from ratio of the carbon dioxide concentration in the air to that in the leaf interior. The rate of carbon fixation is proportional to the ratio of the conductance of the carbon

dioxide flow into the leaf at the stomata, to that at the cell membrane. Respiration is divided into maintenance respiration and growth respiration. Maintenance respiration is dependent on the temperature. It is proportional to the carbon content and dry weight of each compartment considered. Growth respiration is the cost of the conversion of glucose into structural biomass. Growth respiration is calculated from knowledge of the biochemical composition and is estimated as a fixed fraction. When maintenance respiration becomes greater than photosynthesis, the needles fall. Low magnesium content in the needles reduces photosynthesis. Carbon and nutrient allocation is calculated on a daily basis. A pool of non-structural carbon reserves is available for bud flush and repair needs. Respiration needs must be met before assimilates can be allocated to growth or to non-structural carbon storage. The leaves have first access to assimilates. Although the tissue of highest priority is given first opportunity to use the carbon available, growth is limited by a maximum daily growth rate which can be obtained from the allometric relationships that are measured. Nutrient allocation is mainly the opposite of carbon allocation. The fine roots have first access to the nutrients. Allocation is influenced by the environmental conditions, whenever a resource is scarce. The root uptake is modelled mechanistically. Root density, root geometry and root function are accounted for by assigning a volume of soil to the individual root. TREGRO contains a sub-model for the transport and exchange phenomena, occurring at root level (Tiktak, Bredemeier and van Heerden 1995). It takes into consideration air pollutants such as ozone. Ozone is modelled as injuring the mesophyll cells when above a certain level.

FORGRO

FORGRO is a stand level physiology-based carbon-balance model of forest growth. Using a time step of one day, 150 year long succession modelling can take place. The biomass of a stand is represented in the model by the total dry weights of foliage, branches, stems and roots. The root biomass is separated into fine and coarse roots. The model calculates the stand characteristics and the yield of the ground surface area (Mohren, Bartelink, Jorristma and Kramer 1993.). The leaf area index is calculated from the total foliage dry weight. The foliage of conifers is divided into 10 different age classes. Canopy assimilation is estimated from the photosynthesis rate of individual leaf layers inside the canopy. The gross photosynthesis for different leaf layers in the canopy is calculated by use of a negative exponential equation, that describes the relationship between the photosynthesis per unit of leaf surface and the amount of photosynthetically active radiation absorbed. The photosynthesis rate decreases with increasing needle age. Daily canopy assimilation is modelled by integrating the photosynthesis-light response curve over the leaf layers in the canopy and over the day, using a Gaussian integration scheme. Growth respiration is determined by the approach of Penning de Vries, in which three biosynthetic processes are assumed to be associated with growth: the conversion of glucose into organic components, the translocation of assimilates to the growing site, and the reduction of the nitrogen, as part of protein synthesis (Penning de Vries, Jansen, ten Berge and Bakema 1989). The efficiency of conversion from assimilates to dry matter can be calculated from knowledge of the biochemical composition of the structural biomass. In the model it is assumed that maintenance respiration is linearly related to the mineral and the nitrogen content. Maintenance respiration has been found to increase exponentially with the temperature (Penning de Vries *et al.* 1989). The dry weight of the stem is converted into stem volume, using the average density of stemwood. The increment in stem diameter is calculated from a taper function or genuine volume function, using such parameters as form, height and volume increment of the stem. Phloem and bark are incorporated into the estimate of the dry weight of the stem. The tree characteristics that are calculated are for an average tree in a forest stand. The ability for thinning is also incorporated into the model. The photosynthetic allocation to various biomass components and tissues is an important process in whole-tree carbon balance models. Allocation to various parts of the tree is simulated by use of an empirical distribution key. Fixed sets of assimilate distribution functions are applied. Forest hydrology is included in the model through keeping

track of the soil moisture on a day-to-day basis, distinguishing between different soil layers. The soil moisture is depleted by root uptake, which in turn is determined by transpiration, and it is replenished by infiltration of the net precipitation. Leaching to deeper soil layers occurs when the soil water content exceeds the field capacity. Uptake by roots is decreased when soil moisture content approaches the wilting point. Interception loss is estimated by subtracting an amount of canopy interception determined by the leaf area index, LAI. Transpiration is determined using a Penman-Monteith evapotranspiration equation, the stomatal conductance being derived from the net photosynthesis and the vapour pressure deficit of the air, and foliage water status being determined by the combination of water loss through transpiration, soil water availability and tree water uptake. The effects of gaseous air pollutants are modelled on the basis of experimental results concerning their influence on physiological process rates. Minimum and maximum requirements for the uptake of nutrients are calculated by taking into account the minimum requirements of new tree tissue that is formed and of reallocated amounts of dead tissue nutrients. Nutrient uptake is modelled with help of the nutrient concentration in the soil, the root architecture and the root functioning. Transport and exchange phenomena are modelled at the rhizosphere level. FORGRO has been applied to the German Solling spruce site for evaluating the effects of air pollutants on spruce forests (Mohren *et al.* 1993.).

Forest-BGC

FOREST-BGC (Forest-Bio-Geo-Chemical simulation model) is a model developed by Steven Running and collaborators (Running and Gower 1991). It originated as a stand-level model of forest biogeochemical cycles, and is designed for multi-scale applications to explore ecosystem interactions. The model has both daily and annual time steps, recognising differences in the responses of different ecosystem processes. Hydrologic and canopy gas exchange processes are calculated daily, whereas

ecosystem carbon and nitrogen processes are computed annually. Different tree species are not explicitly identified, although species-specific physiological characteristics can be represented. Geometric complexities of different tree canopies are reduced to simple quantification of the sum of all leaf layers as the leaf area index (LAI). Internal tree physiological processes such as carbon and water transports are simplified. The below-ground root system and soil processes are also treated in a simplified way to reduce the requirements on data. Only standard meteorological data, such as daily maximum and minimum temperature and precipitation are needed. Some parts of FOREST-BGC has been thoroughly tested, particularly the hydrologic and carbon cycle components (Korol, Running and Milner 1995). An interesting aspect of model, which it shares with the next model PnET, is the existence of extended versions of it, in which the model has been upscaled in time, viz. BIOME-BGC (Running and Hunt Jr. 1993) and the spatially scaled RHESSys (Band, Patterson, Nemani and Running 1993). The spatial up-scaling in particular makes it possible to apply the model at a landscape level, with broad-scale data input. A global model has also been derived from the FOREST-BGC model: GESSys (Waring and Running 1998).

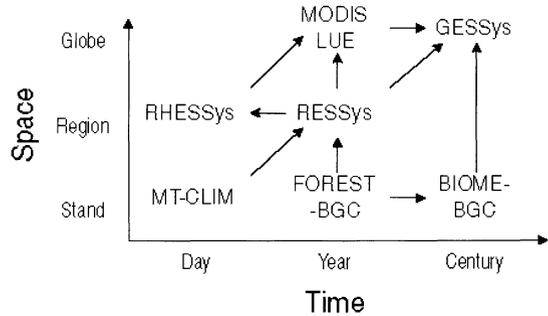


Figure 5.6: *The BGC family of ecosystem simulation models.*

PnET

PnET is a lumped-parameter model of carbon and water balances in forests. It was developed on the basis of two principal relationships: the maximum photosynthetic rate is a function of the foliar N concentration, and the stomatal conductance is a function of the realised photosynthetic rate (Aber and Federer 1992). Monthly LAI, and carbon and water balances are predicted by combining these relationships with standard equations describing the light attenuation in canopies and photosynthetic response in diminishing intensities of radiation, along with the effects of soil water stress and of vapour pressure deficit. The model is developed in Visual Basic and has a user-friendly interface. Several versions of PnET has been developed for different purposes: PnET-Day, PnET-II, PnET-CN, PnET-CHESS, whereof the last three mentioned operates with a monthly time-step, and have no specified spatial dimension. CHESS is a soil chemistry model developed by Santore and Driscoll (1995). The soil processes in PnET are represented in a simplistic way. PnET-Day uses foliar mass, specific leaf weight, foliar N concentration, temperature and radiation flux to predict daily gross and net photosynthesis of entire forest canopies. It has been validated against daily summaries of eddy-correlation carbon-balance measurements from the Harvard Forest (Aber, Reich and Goulden 1996). PnET-II adds carbon allocation and respiration terms, as well as a full water balance to predict NPP, transpiration and runoff. Empirical soil respiration terms permit the prediction of the total ecosystem carbon balance under ambient conditions. This version has been validated against annual NPP and monthly water yield data from the Harvard Forest and Hubbard Brook ecosystems and is used to predict the combined effects of climate change and increased atmospheric CO₂ on these processes (Aber, Ollinger, Federer, Reich, Goulden, Kicklighter, Melillo and Richard G. Lathrop 1995). PnET-CN adds compartments for woody biomass and soil organic matter, as well as algorithms for biomass turnover and of litter and soil decomposition to permit the calculation of complete carbon and nitrogen cycles. This version maintains the predictions of NPP and water balance used for the validation in PnET-II, and also compares well with field data in predicting the total annual, mean seasonal, and actual time series rates of nitrate loss in streams (Aber, Ollinger and Driscoll 1997, Aber and Driscoll 1997). An additional version of the model (PnET-BGC) is under development.

FORSANA

FORSANA is a tool for estimating major pools and fluxes of carbon, nitrogen and water within a forest, including the trees, the ground vegetation and the rooted soil layers (Grote and Erhard 1999). Only broadly available forest inventory data are required as initial stand variables. For the soil data, information about the profile is needed, which can be obtained from soil mappings or on the basis of standard soil properties for specific types of soil profiles. Daily climate data are used as driving variables. Nitrogen deposition and air concentrations of SO₂ can also be used as optional inputs. The forest stand itself is assumed to be homogeneous horizontally. All information is scaled upwards from average stems to the stand area. The canopy and the rooted soil are divided into horizontal layers, characterized by their specific leaf and root mass, respectively. For the soil, the thickness of the layers is derived from the soil profile of the site, whereas the canopy layers are of equal thickness. The aboveground parts of the herbaceous vegetation are modelled as a separate layer of foliage beneath the canopy and share a number of initialised soil layers with the tree roots. FORSANA was developed within the framework of the project SANA ('Sanierung der Atmosphäre über den neuen Bundesländern'), which was launched in 1993. The daily processes were verified by field measurements from three sites, different in age and pollution history, the verification of stand development was done with the use of inventory data (Grote, Erhard and Suckow 1997).

5.3 The Integrated ForSAFE model

The review we concluded of different models led to many new ideas and inspiration concerning how to build the ForSAFE model, the model specially tailored for many of the SUFOR Programme's specific questions. It was realized that present models either focus on growth and biotic processes, largely ignoring geochemistry and soil chemical conditions, or focus on soil chemistry and tend to treat growth and biotic processes superficially. None of the models surveyed actually had a good balance, and many of them were used by simply calibrating essential parameters. Both the lack of balance between foci and the somewhat uncritical calibration routines were felt to be a problem and to be a severe obstacle for a-priori predictions based on first principles. Thus, the decision was made to develop ForSAFE. Different version of the PnET and FORSANA models in particular were useful, since the source codes were easily available and the authors of these models were very open to cooperation and to discussions on further developments. The philosophy was to take good parts of these models and combine them with the best parts of our own models, fitting them together into a single soil-tree model system. Some parts were missing and these needed parts had to be developed specifically for ForSAFE.

The ForSAFE model consists of several sub-models, some of which have existed for some time as separate models, particularly the soil chemistry model SAFE. Models currently being developed are a browsing model, the decomposition model for soil organic matter DECOMP, a model for growth and a soil profile hydrological model based on a successful earlier model, the PULSE variant of the HBV model (Bergström 1991). The following is a list of the sub-models contained in ForSAFE:

1. The growth sub-model:
 - (a) Tree growth (principles taken from PnET (growth drivers) and FORSANA (allocation to tree compartments, age cohorts and stem diameter classes)): for stemwood, leaves, litterfall, and roots
 - (b) Nutrient allocation (principles taken from PnET and FORSANA)
 - (c) Uptake processes for base cations (Sverdrup-Warfvinge critical uptake concept, including the BC/Al retardation functions)
 - (d) Uptake processes for nitrogen
2. SAFE:
 - (a) Ion exchange
 - (b) Soil solution equilibria
 - (c) Chemical weathering
 - (d) Soil nitrogen transformation processes
 - i. Denitrification: Sverdrup-Ineson concept
 - ii. Nitrification: Michaelis-Menten kinetics according to Sverdrup-Warfvinge
3. DECOMP:
 - (a) Decomposition of organic matter
 - (b) Release of nitrogen
 - (c) Release of dissolved organic carbon
 - (d) Nitrogen immobilisation
4. The stress sub-model:
 - (a) Soil acidity: BC/Al concept

- (b) Ozone, dose-response regressions, AOT
 - (c) Root rot
 - (d) Frost
 - (e) Drought
 - (f) Insects and pests and their effects
5. The biodiversity models (FORSAFE-VEG, SUMO-MOVE, BIOSYS): A general biodiversity model is envisioned, but the basic concept and the scope of the model are still under development. The model will take account of insects found on dead wood, tree species found in the landscape and the ground vegetation classes that may be functional.
 6. The hydrological model for a soil column with optional number of layers adapted from the HBV/PULSE-model
 7. The wildlife-browsing model: A forest browsing model which includes natural regeneration
 - (a) Regeneration feedback
 - (b) Fauna population dynamics and impacts of hunting

The decomposition of soil organic matter is an important process that contributes to the soil development processes and to the acid-alkalinity balance in the soil, besides being of great importance for the recycling of carbon, nitrogen and nutrient salts in the ecosphere. The processes involved have important implications for such matters as global climate change, forest management practice, soil podzolisation, acidification processes and soil conservation. Integrated soil chemistry models for calculating soil acidification and lake, stream, groundwater and forest soil acidification sensitivity have been developed for the Swedish Environmental Protection Agency. The steady-state PROFILE model (Warfvinge and Sverdrup 1991, Sverdrup and Warfvinge 1992) is being used as one of the tools to map critical loads for streams and forest soils in 15 different European countries. SAFE, the dynamic version of PROFILE, is being applied to different Swedish field research sites, for assessing the effects of future acid deposition scenarios. SAFE will be modified through biomass decomposition and forest growth processes being incorporated into it in greater detail, so as to describe the effects of acid deposition, forest management practices, podzolisation and soil genesis on the ecosystem. SAFE and PROFILE differ from all other models in the weathering rate and the production of base cations from this process not being subject to the usual calibration procedures, but being calculated explicitly from the soil mineralogy and first principles. Several earlier soil and water acidification models, including a soil biomass decomposition module, are known: ILWAS (Chen, Gherini, Hudson and Dean 1983) and its derivate NuCM (Johnson, Swank and Vose 1993, Johnson, Binkley and Conklin 1995), RESAM (de Vries and Kros 1989) and SOIL-N (Jansson 1991). The present approach is based on the Walse-Berg-Sverdrup concept (Walse 1998) and several on new developments.

Developing the dynamic forest ecosystem model ForSAFE is a process that will comprise five major steps, the first two of which have already been carried out. The different steps are:

- Finding a suitable forest growth model.
- Merging it with the DECOMP model.
- Adding new models and changing undesirable sub-models in the forest growth/DECOMP-model.
- Merging the forest growth/DECOMP model with the SAFE model.
- Adding to sub-models for e.g. economic calculations and wildlife influence.

After a thorough survey of several different forest growth models, various of which were mentioned earlier in the chapter, the choice was made for the PnET-family of models, also described earlier in this chapter. The CN-version (where CN stands for carbon and nitrogen) of PnET satisfies most of the conditions above. PnET is based on the assumption that maximum photosynthesis, which in itself is driven by climatic factors, of course, is a function of the foliar nitrogen content. The PnET-model uses foliar nitrogen concentration to calculate maximum instantaneous rates of gross and net photosynthesis which are then reduced for suboptimal temperature, radiation, and vapour pressure deficit (VPD). Since the CN-version also has closed loops for both carbon and nitrogen, as well as water, it clearly meets the first criterion. The temporal scale of a model that is to simulate over a time-horizon, such as in the SUFOR-project, is very important. Too short a simulation time-step requires large, cumbersome and hard-to-get indata files, the actual runtime of a simulation being, of course, much longer. There are climate generation models that generate daily climate data files from monthly average values, but it seems pointless to introduce still more uncertainties in a model, aggregation analyses also suggesting that mean monthly climatic data will do nicely (Aber *et al.* 1995). On the other hand, if the time-steps are too long, the model does not consider even seasonal dynamics, something that undoubtedly will be important in future climate change scenarios. Accordingly, a monthly time-step was considered appropriate for the task. On a spatial scale, PnET is designed for the stand-to-small-watershed level. There is also a version of PnET that includes spatial distribution, but since there is no connection between two contiguous grids, these do not affect each other. PnET requires monthly average data on maximum and minimum temperature and monthly sums of precipitation and photosynthetically active radiation (PAR). PAR, or even global radiation, is not a standard measure at most weather stations. Nevertheless, PAR can easily be derived from global radiation measurements (Aber and Freuder 2000), and global radiation can be estimated fairly well from sunshine duration (Grote and Suckow 1998). Slightly less than 50 parameters were needed to run PnET-CN in its original version. A few of them are species-specific, others are site-specific and still others are both site- and species-specific. Many parameters, however, are specific for a forest type, deciduous or coniferous (Aber *et al.* 1995, Aber *et al.* 1997). The site- and site/species-specific parameters are usually fairly easy to obtain or derive from general forest inventory data. The core of PnET is largely based on recognized physiological relationships and processes, which makes it transparent and relatively easy to grasp. Since the disturbance and climate/pollution change scenarios are critical to PnET-CN, care should taken in developing them for a new site. When this crucial work is done, the model works uncalibrated, however. PnET is an open model, the results, data files, compiled program files and source code are all being made available free of charge to anyone interested in working with or modifying the model. The model can be downloaded from, and the PnET statement of open development can be read on the Internet, <http://www.pnet.sr.unh.edu/>.

5.3.2 Merging PnET, SAFE and DECOMP

As mentioned before, there are several ways to model forest growth. One way is the old statistical way in which one look at a number of species- and site-specific variables and compares them with growth records. Such modelling methods work fairly well when the climate and soil conditions are constant. They cannot take account of climate change, nutrient depletion or increasing deposition however. Another way to model forest growth is the physiological way, in which one tries to describe every process in the plant in detail. This kind of model is usually very complex, difficult to grasp, slow to run and needs much detailed input data. A third, and in the present case preferred, way of modelling forest growth is a so-called hybrid model. A model of this kind combines the two models just mentioned above without having their disadvantages. This is driven by statistical or empirical data and the predicted growth is based on the simulation of the, for the purpose, most important processes. This provides a model with a believable foundation that is still flexible in the face of change (Kimmins 1997). A forest growth model is also a question of scales, both temporal and spatial, see Fig. 5.6. The rates of the processes in a forest

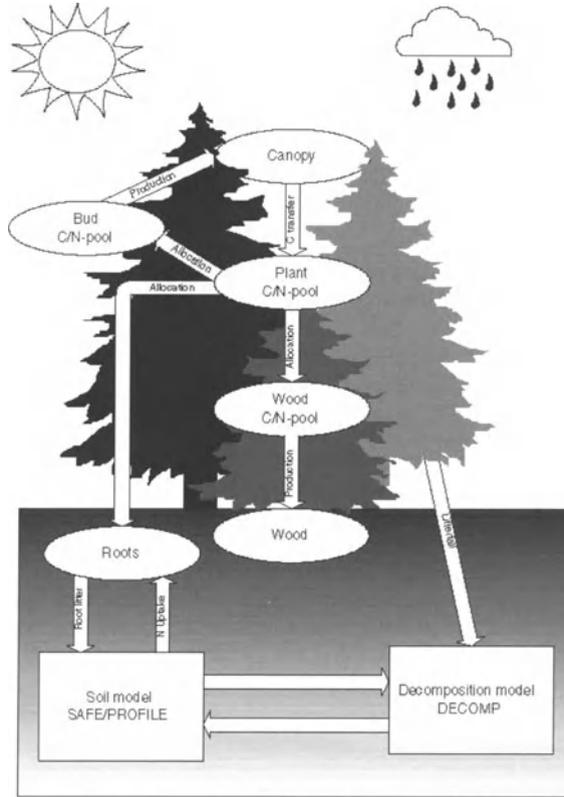


Figure 5.8: A conceptual, very simplified diagram of the forest growth model structure with the interacting models coupled. The arrows show flow paths for carbon and nitrogen. The flow paths for water are omitted for reasons of lucidity

ecosystem vary, depending on the spatial level one looks at, from minutes for the photosynthesis rate to decades and even centuries for large-scale changes in a forest landscape. Thus, a forest growth model has to be multi-scaled in both time and space (Waring and Running 1998).

As a first step in the construction process of ForSAFE, PnET and the decomposition model mentioned earlier developed by Walse *et al.*, henceforth termed DECOMP, were merged into a single model, termed PnET-SUFOR. Since decomposition in itself is a very important part of the carbon and nitrogen cycles in a forest ecosystem, the two models were not combined or linked to each other, but DECOMP was actually programmed into PnET-CN as a subroutine. Since DECOMP was designed to be integrated in the SAFE soil chemistry model, it has many simplifications in common with SAFE. The most important are (Alveteg 1998):

- The soil is considered to be a series of continuously stirred tank reactors in which each tank reactor represents one soil layer.
- It is assumed that all changes in physical and chemical characteristics with depth are discrete.
- The water flowpath is assumed to be downwards only.

DECOMP require, as it stands at present, four different input variables: soil pH, soil temperature, soil moisture content and litterfall. In order to not increase the amount of input data needed, a goal was to have the climatic input used in PnET somehow generate input data to DECOMP. PnET is a forest ecosystem model that concerns mainly aboveground processes, without any soil chemistry being built in. At present, the soil pH and solution chemistry of all four soil layers need to be found in the input data file together with the climatic data. In ForSAFE, the soil chemistry is calculated using SAFE. In SAFE and DECOMP, all processes are formulated as differential equations. In PnET and some of the other models we have been inspired by, this is not the case, their having been formulated as update equations or difference equations using uniform timesteps. The aim is to eventually have all the process descriptions converted to a differential form for better numerical handling and better generality of the model.

Modelling of ForSAFE internal variables

The driving variables in physiologically based models such as PnET and ForSAFE are the climatic variables. An exposition of the employed approaches that link the three sub-models in ForSAFE to each other is in its place.

Temperature Temperature is important to almost all chemical and biological processes. In the ForSAFE model decomposition and weathering are two important temperature dependent processes. To determine whether the decomposition sub-model is sensitive to aggregating mean daily soil temperatures to obtain mean monthly values, a soil temperature data set with data at five different depths on a half-hourly basis was employed. The half-hour values were aggregated to daily and monthly averages. DECOMP was programmed in FORTRAN95 and was run with all the other input data held constant at typical early average values for Asa Research Park. The results showed monthly calculations to give considerably higher mass-loss rates on a yearly basis than daily calculations did. When the calculations were performed using a daily time-step but employing the monthly average value used every day of that a given month, there were only very small differences between the mass-loss rates generated by the daily and the monthly data. There are numerous soil temperature models available, which calculate soil temperature at a given depth from air temperature (Grote and Suckow 1998). Many of them are complex and heavily laden with parameters, pertaining to physical relationships, most of them using a daily or an even smaller time-step, and are for both these reasons not useful in ForSAFE. A simpler but nevertheless useful way to simulate soil temperature on a monthly basis is to consider the soil temperature to be a damped and lagged reflection of a moving average of air temperature (Sandén and Warfvinge 1992). Mathematically, this can be described as follows:

$$T(t, z) = \frac{\alpha - z}{D \cdot z} \cdot \left(T_t^{air} + (\bar{T} - T_t^{air}) \cdot \frac{z}{\alpha - z} \right) + \frac{\alpha - z}{D \cdot z} \cdot T_{t-1}^{air} + \dots + \frac{\alpha - z}{D \cdot z} \cdot T_{t-\frac{D-z}{\alpha-z}}^{air} \quad (5.1)$$

where $T(t, z)$ is the the air temperature at depth z on day t , \bar{T} is the long-term average air temperature, D is the number of days in a year and α is an empirical constant. The soil temperature is also moderated by the vegetation cover and by the forest floor. The tree canopy and the forest floor protect the soil from excessively high summer temperatures through intercepting solar radiation and through reducing the loss of heat during the winter. Since the water in the soil is the main heat conductor the soil moisture content is very important. During a period of drought, the water content of the forest floor becomes very low, thus insulating the soil beneath. This is especially true if the forest floor consists of porous material, e.g. if there is a thick litter layer and/or a moss layer on top. This is not yet accounted for in the present version of ForSAFE. The influence on soil temperature of a snow cover is accounted for by a routine that adjusts the average air temperature to which the soil is exposed in terms of the thickness of the snow cover. The routine by suggested by Sandén and Warfvinge (1992) is involved here. It assumes that 0.5 cm water equivalents of snow cover insulates the soil underneath totally and

that a thinner snow cover has less damping effect if air temperature is below freezing (since the density of snow varies considerably with the conditions it is difficult to state the actual thickness of such a snow cover, but for typical Swedish conditions this would be around 5 cm of dry snow). If a snow cover is totally missing and the air temperature drops below 0°C, this would mean the soil temperature dropping below freezing, at least in the upper layers, and soil frost forming. This is not taken account of in ForSAFE. However, since the soil temperature data from Asa Research Park show that the soil temperature only rarely falls below freezing, even at 10 cm in depth, this is not considered to be a problem. Temperature records are usually easy to obtain, most of them contain both maximum and minimum temperatures. ForSAFE requires average daytime and nighttime temperatures. These are calculated from maximum and minimum values (Running, Ramakrishna and Hungerford 1987) as follows:

$$T_{air}^{avg} = 0.606 \cdot T_{air}^{min} + 0.394 \cdot T_{air}^{max} \quad (5.2)$$

$$T_{air_{day}}^{avg} = T_{air}^{max} - 0.29 \cdot (T_{air}^{max} - T_{air}^{min}) \quad (5.3)$$

$$T_{air_{night}}^{avg} = T_{air}^{avg} - (T_{air_{day}}^{avg} - T_{air}^{avg}) \quad (5.4)$$

Radiation The sun is the source of energy for trees and for the hydrological cycle. The amount of energy that reaches a given spot on earth is determined by four factors: the solar output, the distance from the sun, the angle to the sun and the length of the day. Global radiation data from weather stations usually comes as the sum of the radiation per time unit, MJ/(m²·day) or MJ/(m²·month). Not all of the radiation that reaches the biosphere is used by plants for photosynthesis, but only the photosynthetically active radiation (PAR) in the wavelength range of 0.4-0.7 μm. The photosynthesis routine in ForSAFE requires radiation inputs as PAR μmol/(m²·s). A conversion equation derived by Aber and Freuder (2000) and Ollinger *et al.*, (1995) is employed. The length of the day needed in these calculations is calculated employing an equation suggested by Penning de Vries and Laar (1982). The length of the day depends on the latitude and the declination of the sun angle, which in turn is a function of the season (Spitters, Toussaint and Goudriaan 1986). There are also routines available for calculating PAR from sunshine duration only, if necessary (Grote and Suckow 1998). The routine for light absorption in the canopy makes use of the Beer-Lamberth law (Monst and Saeki 1953):

$$I_i = I_0 \cdot e^{-k \cdot LAI_i} \quad (5.5)$$

where I_i is the radiation at level i , I_0 is the the radiation at the top of the canopy, k is the light attenuation coefficient and LAI_i is the cumulative leaf area index above level i . The calculation of light absorption in the canopy is implemented in the same way as in the FORGRO model (Mohren 1987). This model separates direct radiation from diffuse radiation and distributes it to sunlit and to shaded foliage, respectively. The light attenuation coefficient is calculated for each fraction of the radiation separately, since these fractions change with total radiation and canopy depth. A cluster factor is introduced since foliage is structured according to the branchiness of the trees, and it is calculated using species-specific parameters (Oker-Blom, Kaufmann and Ryan 1991).

The soil water sub-model Soil moisture content is important for all processes that occur in the soil: biological, chemical and, as mentioned above, physical. The fact that the soil water content affects both the weathering process and the decomposition process, both directly through the mass-loss-rate-regulating relationship parameterized by Walse *et al.* (1998), and indirectly through its effect on the soil temperature, makes it very important to model the soil moisture as correctly as possible. In the original PnET-CN model, the soil is considered to be mono-layered and to have the same characteristics throughout the layer. That might provide a satisfactory picture of forest ecosystems of shallow to moderate soil depth and with bedrock

underneath. In other forest soils, where the soil characteristics can change considerably with depth, a somewhat more detailed hydrology sub-model should be employed. Since the soil water flow is an important factor in the SAFE model too, the idea of implementing a simple vertical hydrology model in ForSAFE seems appropriate. The standard configuration of SAFE normally uses 3 or 4 soil layers, dictated mostly by the properties of the majority of Swedish soils. Since the hydrological properties of the litter layer and of the humus layer are very different from those of the mineral soil, the organic layers should be treated separately from the mineral soil layers.

When modelling soil water movements on a monthly basis one is never able to capture the fast dynamics contained in the water cycle. Two or three days after a rain storm, most Swedish forest soils are drained and are at field capacity again. Since one only has monthly data to work with one does not know whether the precipitation reported came as one big storm involving heavy rainfall or as scattered, light showers during most of the days of the month. One way to deal with this is to apply a pseudo-daily time step, that is to distribute the summarized precipitation evenly to each day of the month, and to then calculate the water balance daily, but this approach fails to reflect the fast dynamics of the water cycle needed in the decomposition sub-model. This is avoided in the ForSAFE model by employing an algorithm where the precipitation falling during a month is distributed over the days in accordance with a pattern established by empirical observation. The exact day in the month of hard rain is not so important as the number of days it occurs and to what extent, as well as the number of dry days. The latter approach showed significantly better performance when simulating with the DECOMP model than the former.

The soil water sub-model employed in ForSAFE is a simplification of the PULSE model (Bergström 1991). Certain relevant terms can be defined as follows:

- *Water holding capacity (WHC) or field capacity*: the soil water remaining after long-term free drainage.
- *Wilting point (W_s^{wp})*: the water content of the soil that plants cannot utilize.

Water transport in a layer only takes place if the water content (W) in the soil layer is greater than field capacity. The change in the water content over time, in view of plant uptake (U), soil evaporation (E) and infiltration by percolation from previous layer, (P_{in}), is described by a differential equation:

$$\frac{d}{dt}W = P_{in} - U - E - P_{out} \quad (5.6)$$

where the downward percolation P_{out} is:

$$P_{out} = P_{in} \cdot \left(\frac{W - W_s^{wp}}{WHC - W_s^{wp}} \right)^\beta \quad (5.7)$$

β is an empirical model constant. E being zero if the layer is further down in the profile than the assumed evapotranspiration depth, z_E . Infiltration by percolation from layers above is assumed zero at temperatures below 0 °C. When the air temperature is negative, precipitation is stored as water equivalents in a snow pool. The release rate from this pool is a function of the average air temperature (Weise and Wendling 1974).

$$W_{snow}^{max} = T_{air}^{avg} \cdot (0.45 + 0.1 \cdot T_{air}^{avg}) \quad (5.8)$$

Melt water is added to the uppermost soil layer. If the precipitation beneath the canopy is insufficient to meet the demands of potential evaporation, water down to a certain depth is taken up from the soil. Water vapour pressure deficit, VPD , is needed in order to calculate the evapotranspiration.

$$VPD = VP^{sat} - VP \quad (5.9)$$

The saturation water vapour pressure in the air, VP^{sat} , is calculated from (Murray 1967)

$$VP^{sat} = 6.1078 \cdot e^{\left(17.2694 \cdot \frac{T_{air_day}^{avg}}{(T_{air_day}^{avg} + 273.3)}\right)} \quad (5.10)$$

There are several methods of calculating the the evapotranspiration: hydrologic, climatological, micrometeorological and combinations thereof (Rosenberg, Blad and Verma 1983). Both the Penman-equation, which is a combined method, and a modification of it by Montieth are widely used (Rosenberg *et al.* 1983). It is not trivial to employ this type of equation (Bengtsson 1997). Many equations of this sort have been derived and parameterized for agricultural crops of fairly homogeneous character (Ventura, Spano, Duce and Snyder 1999), Forest are generally more heterogeneous than this and the parameterization of them is difficult (Rosenberg *et al.* 1983). A simpler approach is used in the PnET-model family (Aber and Federer 1992, Aber *et al.* 1996). A constant fraction of the precipitation is intercepted and evaporates; this includes both rain interception evaporation and sublimation from snow in the canopy and at the ground level. The rest is put into the available soil water pool. Transpiration is calculated on the basis of water use efficiency (mg carbon fixed per g transpired water), WUE . WUE only varies with CO_2 and with the water vapour concentrations in the air. If the CO_2 concentration is regarded as constant, it becomes a function of the vapour pressure deficit only. This provides a direct connection between carbon gain and transpiration.

$$WUE = \frac{WUE_{const}}{VPD} \quad (5.11)$$

WUE_{const} is found to be 10.9 for deciduous trees (Aber and Federer 1992) and 7.3 for *Picea abies* (Kram, Santore, Driscoll, Aber and Hruška 1999). The potential daily transpiration is calculated as:

$$PotTrans_d = \frac{PotGrossPsn}{WUE} \quad (5.12)$$

From a daily water balance that is calculated, a water stress factor is derived:

$$\begin{aligned} Trans_i &= PotTrans_d \text{ when } Water_i \geq \frac{PotTrans_d}{f} \\ Trans_i &= Water_i \cdot f \text{ when } Water_i < \frac{PotTrans_d}{f} \end{aligned} \quad (5.13)$$

where $Water_i$ is plant available water day i of the month and f is a soil water release factor (the fraction of plant available water that can be removed on one day without causing water stress). A water stress factor used in the calculations of photosynthesis (see eq. 7.32) is calculated as:

$$D_i^{water} = \frac{Trans_i}{PotTrans_d} \quad (5.14)$$

This very handy and user-friendly solution to the water balance problem is implemented in the ForSAFE model. Since interception depends on the species, age and stand density (Kimmins 1997) the interception storage capacity should be calculated dynamically. An interesting approach, based on the morphological variables leaf area index (LAI) and crown area index ($CRAI$), to predicting rainfall interception, suggested by Liu (1997), and is being considered for inclusion in the ForSAFE model.

$$Int = W_{int}^{spe} \cdot LAI \cdot \left(1 - \frac{W_{int}}{W_{int}^{spe} \cdot LAI}\right) \cdot \left(1 - e^{\left(- (1 - F_{thr}) \frac{prec \cdot CRAI}{W_{int}^{spe} \cdot LAI}\right)}\right) \quad (5.15)$$

where W_{int}^{spe} is the specific canopy interception capacity, for *Picea Abies* 0.34 mm/ LAI , as cited in Eckersten *et al.* (1995), and W_{int} is the already wetted fraction of the canopy. F_{thr} is the fraction of throughfall. Since ForSAFE is run on a monthly time step, using the sum of one month's precipitation as input data, the inter-monthly distribution of precipitation is needed for the calculation of W_{int} . Thus, the waterbalance is calculated on a pseudo-daily basis.

5.3.3 The SAFE model and soil chemistry

SAFE is based on a conceptual model of a forest soil, represented either as a profile or as the whole catchment. SAFE is closely linked to the PROFILE model, PROFILE being the steady-state version of the dynamic model SAFE, which takes the gradual changes in soil state over time into account.

SAFE contains the following chemical subsystems:

- The deposition, leaching and accumulation of dissolved chemical components
- Chemical weathering reactions of the soil minerals with the soil solution
- Cation exchange reactions
- Reactions of the nitrogen compounds: nitrification and denitrification
- Nitrogen uptake
- Base-cation uptake
- Solution equilibrium reactions involving carbon dioxide, aluminum species and organic acids

These processes of course only represent a selection of the chemical reactions in the soil. Among the processes that have not been included are sulphate adsorption and, a series of reactions that can change the CEC of the soil matrix, store sulphur irreversibly or affect the ANC balance in certain soils. All the processes included in the model have been subject to a necessary simplification in some respect. The obvious principle is to divide the soil into compartments that correspond to the natural soil stratification, as the soil horizons are the largest chemically isotropic elements in the system. This stratification results in a marked difference in chemical properties between the different soil layers.

5.4 Summary

The three stand-alone models PnET-CN, DECOMP and SAFE are being merged into a new model termed ForSAFE. Some of the connections between the models are shown in Fig. 5.8. The same climatic drivers as used in PnET are used to calculate the necessary soil temperature and soil moisture content employed in both the DECOMP and the SAFE modules. PnET provides both above- and belowground litterfall inputs to DECOMP. DECOMP calculates the decomposition of the soil organic matter and the available nitrogen for uptake by the trees in PnET. SAFE provides pH and base cation concentrations to DECOMP and to PnET for uptake.

Chapter 6

Models of the risk of windthrow and frost

Kristina Blennow, Mattias Carlsson, Erik Johansson and Ola Sallnäs

6.1 Introduction

Several hazards for forest production can be identified, such as damage caused by wind, frost, insects or pathogens. Modelling efforts of ours have been concentrated upon the risks of damages by wind and frost, landscape level models for which will be presented below. A landscape is defined as an area of anywhere from a few km² to several tens of km².

6.2 The risk of windthrow

Windthrow is one of the major disturbance factors in south Swedish ecosystems and although it is considered a hazard in terms of forestry production, it has positive implications for biodiversity. Wind and snow cause damage to production in Sweden corresponding to 4 million m³ each year, representing an annual cost of about 150 million EUR (Valinger and Fridman, 1997).

During two recent storms, on November 30, 1999 and December 3, 1999, about 5 million m³ of forest was windthrown (damages of 200 million EUR) in southern Sweden alone (Swedish Board of Forestry, 1999). As shown by the figures just cited, windthrow varies both in time and in space. Consequently, the proportion of damage done to a particular forest estate may be much larger than the average figures for large areas indicate. In future regional climatic scenarios for Sweden, the windiness in many places increase significantly as compared with the present climatic conditions (Blennow et al., 2000), so that the risk of windthrow can be expected to increase. Our risk assessment model involves several components. It is designed for assessing

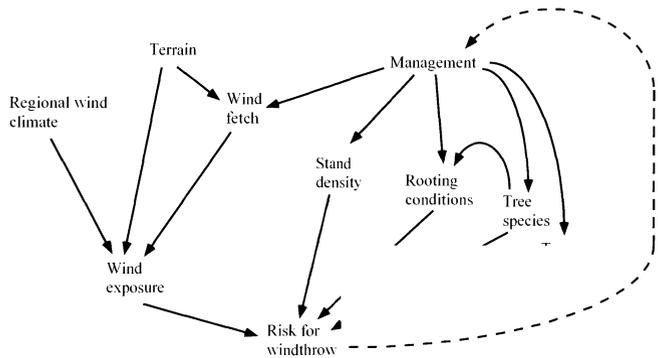


Figure 6.1: A causal loop diagram showing the relationship among the factors affecting the risk of windthrow.

Our risk assessment model involves several components. It is designed for assessing

the risk of windthrow at a landscape level under conditions such as those found in southern Sweden. In its present configuration, the model calculates the risk of windthrow at at least 10 m high edges of wind-exposed stands. In southern Sweden, the topography of the ground surface is gentle and, most of the forest stands are small (a few ha to tens of ha). Norway spruce, Scots pine and birch are currently the most frequent tree species. In the recent storms described above, the wind damage was concentrated to the edges of stands, to stands that were recently thinned, to seed trees and to shelter-wood (Swedish Board of Forestry, 1999). In the area in question, strong winds often occur connected with the occurrence of cyclones, which is a large-scale weather system. We assume that within a given landscape the weather is basically the same, i.e. caused by the same weather system. Such large-scale weather events are modulated, however, by the terrain. Consider a situation involving strong wind. The wind varies across the landscape, mainly as a result of variations in the terrain. The terrain includes not only the topography of the ground surface but also the forest cover. Thus winds are affected by the forest conditions (Fig. 6.1); a tall forest is aerodynamically rougher, for example, than open ground. The stability of a stand is also affected by forest management measures, such as the choice of tree species and the thinning regime (e.g. Persson, 1975; Lohmander and Helles, 1987; Valinger and Fridman, 1997; Jellesmark et al., 1998; Peltola et al., 1999; Gardiner and Quine, 2000). If the wind load is greater than the resistance of the trees, trees will uproot or break. The risk of windthrow is the probability of the wind load exceeding the stability of the trees.

6.2.1 Model

The input data to the model designed for assessing the risk of windthrow consisted of forest inventory data and a digital elevation model (DEM) stored in a geographical information system (GIS) for the area and surroundings, as well as data on climate from a nearby meteorological observation station (Fig. 6.2). Data on the forest cover

(tree species, tree height, stem diameter at breast height, and number of stems per ha), which could be obtained from the landscape projection model, was the input to a model component that identified exposed edges of stands and parameterised the aerodynamic properties of the terrain (roughness length and zero-plane displacement height). Calculation of the aerodynamic parameters was carried out in accordance with the GALES model (Gardiner et al., 2000) and is based on the work of Mayhead (1973) and of Raupach (1992; 1994). Weightings of these

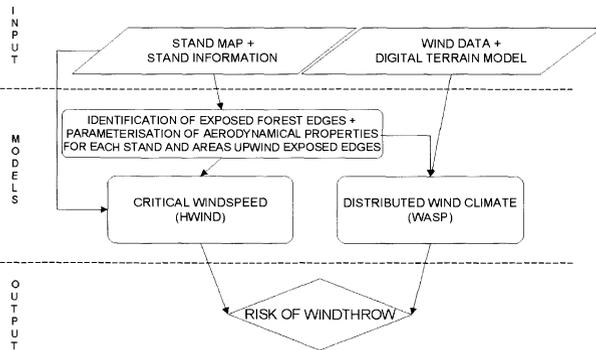


Figure 6.2: Calculation sequence in the model for assessing the risk of windthrow. (Blennow and Sallnäs, 2000).

were made (Mortensen et al., 1998) in front of exposed edges where also calculations of the wind fetch were made. These weighted parameter values and wind fetch values together with forest cover data served as inputs into the Finnish model HWIND (Peltola et al., 1999), that was used for the response component. This component concerns the stability of the exposed edges of a

stand in terms of the wind speeds that are critical for stem breakage and up-rooting. The model includes routines for handling the effects the immediate surroundings have on the wind speed at the edge of a stand (see description of exposure component below). HWIND was modified to handle as variables the parameterised aerodynamic properties of the terrain in front of exposed edges. The critical wind speeds are given as one-hour averages. In calculating these values, the model takes account of the gustiness of the wind, using empirical relationships established in wind tunnel experiments (Gardiner et al., 1997). The Danish exposure model Wind Atlas Analysis and Application Program (WASP) was used to distribute the wind climate over the terrain within the landscape. Climatic data on winds that was obtained from a nearby meteorological observation station, together with data on the topography and the parameterised aerodynamic properties of the landscape that was studied and of its surroundings was fed into the program. The risk of windthrow is the probability the critical wind speed for uprooting or stem breakage that was calculated being exceeded.

6.2.2 Evaluations

Data from two cases of windthrow in the Asa Experimental Forest that occurred in January 1993, were used to evaluate the model (Blennow and Sallnäs, 2000). The windthrow observed was compared with the windthrow predicted for the corresponding wind conditions and state of the forest cover. The modelled distribution of wind speeds in the Asa area is partly a function of the wind direction (Fig. 6.3). The block diagram is overlaid with modelled values of the zero plane displacement height, d . d is used as a zero plane for modelling of the wind field over rough surfaces, such as the forest terrain, and is dependent on the aerodynamical properties of the ground. Roughly, it corresponds to 70-80% of the height of the vegetation. Stacked on top of the block diagram: Two contour maps of the generalised modelled windspeed at 10 meter above the zero plane d for two windthrow occurrences in January, 1993. (Blennow and Sallnäs, 2000.) The terrain also has a strong influence such as shown in speed-up effects over the windward side of the ridge west of lake Asasjön during conditions of westerly wind. In a ground-based field survey, windthrow was observed in 39 of the 310 forest stands altogether. The numbers of windthrown trees per stand varied between 1 and 502 (0.1-13.5%) (Fig. 6.5). In the field survey, no discrimination was made between uprooting and breakage. Since the model in its present configuration only tests for windthrow at the exposed edges of stands at least 10 m in height, only 199 of the 310 stands were tested. Windthrow was observed in 32 of these stands. The model predicted windthrow to occur in 21 of the stands, 12 of which were ones in which windthrow was observed. These preliminary results of the evaluation show good agreement between predicted and observed windthrow (Fig. 6.5).

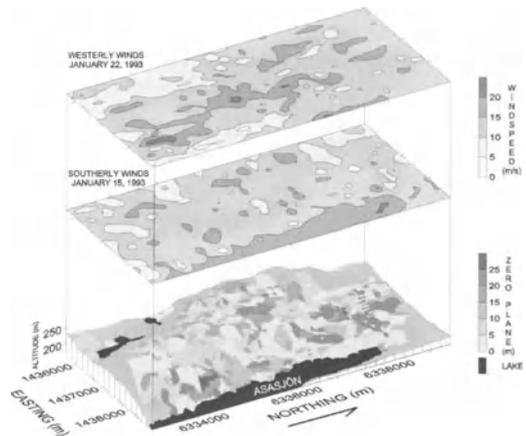


Figure 6.3: *Bottom: Block diagram of the terrain in Asa Experimental Forest and immediate surroundings. Stacked on top of the block diagram: Two contour maps of the generalised modelled windspeed at 10 meter above the zero plane for two windthrow occurrences in January, 1993.*

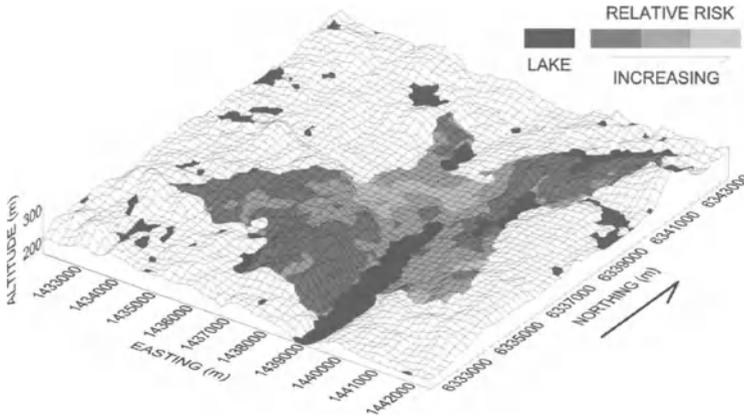


Figure 6.4: Digital elevation model covered with model estimates of air temperature in 3 classes at 2 m above clear felled ground. The model estimates the air temperature for an area of a size corresponding to the size of the forestry units, had one such area been cleared of trees while surrounded by remaining forest.

6.2.3 Discussion

The preliminary results just referred to show rather close agreement between predicted and observed windthrow. A sensitivity analysis of the model will be carried out. The model will also be coupled more closely to the landscape projection model, making it possible to explore whether and to what extent the risk of windthrow can be reduced and what it would cost. Tentative management measures for reducing the risk of windthrow are the appropriate choice of location, of species and of thinning regime. The model will be used to quantify and evaluate management measures in terms of the risk of windthrow under changed climatic conditions (Blennow et al., 2000).

6.3 The relative risk of frost

Frost (temperature below 0 °C) during the growing season can be critical for the bud development, regeneration and survival of plants (Sakai and Larcher, 1987; Blennow, 1998; Blennow and Lindkvist, 2000). Frost also reduces growth and may act as a stress agent, predisposing plants to other types of damage than cold-related damage and affecting the competitive capacity of the plants (Langvall et al., accepted). During the growing season in southern Sweden, low night-time temperatures, frost included, can develop close to the ground due to radiative cooling (Blennow, 1993). On such occasions, the weather is often dominated by an anti-cyclone that promotes radiative cooling and a low degree of mixing of the layers of air close to the ground in the case of clear skies and of low windspeed, respectively. Within a given landscape, some parts of the terrain are more prone to low temperature than others, since variations in topography, ground cover and soil conditions affect the energy balance, resulting in strong variability in the temperature close to the ground (Geiger, 1965). On clear and calm nights, this spatial temperature variability is of a magnitude sufficient to have significant differences in biological impact. In a complex terrain, spatial temperature variability on a micro-scale may even over-shadow effects on a local scale (connected with variations with a horizontal extent of anywhere from 100 m to 10 km) (Blennow and Lindkvist, 2000). Models have been constructed to enable different parts of the terrain to be differentiated in terms of the effects of frost on the early development

of forest stands (relative risk) (Blennow and Persson, 1998; Söderström, 2000).

6.3.1 Model

The modelling methodology employed, developed by Blennow and Persson (1998), is applied here to the Asa area (Söderström, 2000). It involves fitting a linear regression model to spatially distributed temperature data, using independent variables derived from a GIS. The GIS includes a DEM and information regarding the tree cover. Air temperature data of high spatial resolution used as the dependent variable, were collected 2 m above the ground from a moving road vehicle. For technical reasons, meteorological data were collected within a limited part of the study area only (Asa Experimental Forest).

Although the predictor variables per se are basically independent of the weather conditions they influence the spatial pattern of temperatures on clear and calm nights. On such nights, the air temperature generally increases with an increase in height above the ground (inversion). Consequently, altitude was tested in the model as an independent variable. In analogy with catchment area in hydrology, the percentage of the area of a 9x9 pixel moving window that potentially contributes cold air by means of drainage to the central pixels was also tested as a variable (C) in the model. This variable was calculated from the DEM. After resampling, each pixel in the DEM corresponded to an area of 125x125 m on the ground. The drainage of cold air drainage due to gravity is less pronounced, however, in flat areas in which the air may be stagnant. Large differences in elevation also enhance the dynamic turbulence, diluting the skin layer of cold air close to the ground (Kalma et al., 1992). For studying the influence of these features, use was made of a measure of the local relief, expressed as the maximum difference in altitude found within a radius of 250 m. The distance to the large body of water that was closest was also tested as a variable, as was soil type, expressed in terms of the presence of sediment or till. In modelling the air temperature inside a forest, the influence of the forest on the production of cold air need to be considered. Surfaces that are exposed to the cold night-sky cool because of net radiative energy losses which are followed by cooling of the air. In open areas, this leads under calm weather conditions to the development of a ground inversion. For forests, however, most surfaces exposed to the sky are located some distance from the ground,

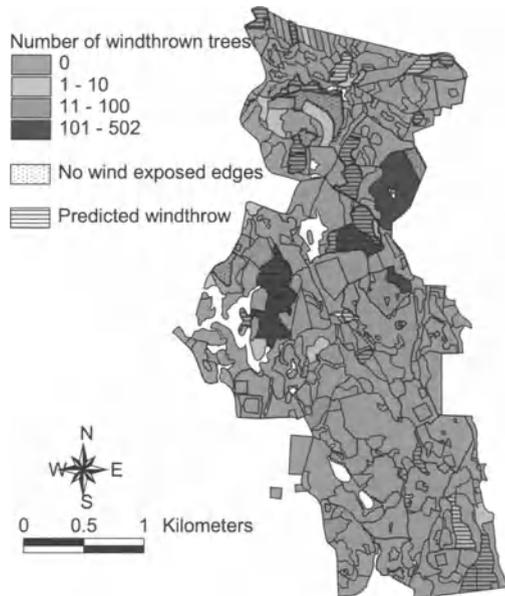


Figure 6.5: *Forest stands in Asa Experimental Forest with the number of windthrown trees marked, as observed in a ground-based field survey after two windthrow events in January, 1993. In addition, those forest stands are marked out for which windthrow was predicted by the model, as are those with no wind exposed edges identified. (Blennow and Sallnäs, 2000.)*

For forests, however, most surfaces exposed to the sky are located some distance from the ground,

which results in there being active surfaces that are elevated. Air which is cooled in contact with such surfaces sinks due to cold air being of higher density. This leads to an increase in mixing and to a less degree of stratification of the air inside the forest than that over open ground (Odin et al., 1984; Kaimal and Finnigan, 1994; Blennow, 1998). The radiation exchange that takes place between the forest floor and the sky depends on the height and the density of the forest and can be parameterised by the sky view factor (Reifsnyder, 1967). Dozier and Frew (1990) developed an algorithm for the calculation of the sky view factor from a DEM used in GIS applications. By use of this algorithm, the sky view factor was calculated for the floor of a canyon formed by a road that traversed the forest-covered terrain.

6.3.2 Results

The model was able to explain 70% ($p < 0.0001$) of the spatial air temperature variation. It was validated by predicting the air temperature for each measurement point along a separate validation transect which was not included in the model construction. The predicted and measured air temperatures compared well ($R^2 = 60\%$). The distributed output of the model is presented as a block diagram in which areas more prone and those less prone to low temperature and frost after small-scale clear felling are delineated (Figure 5). The diagram provides a comparison of the air temperature 2 m above the ground in clear felled areas of representative size (corresponding in size to that of the individual forest stands) if areas of this size had been felled in different parts of the otherwise undisturbed terrain

6.3.3 Discussion

The model developed here provides a means of classifying the terrain into areas that are more and those that are less prone to frost (relative risk) during the growing season and after felling. As can be seen in Fig. 6.4, flat and low-lying parts of the terrain are those areas most prone to low temperature and frost after felling. In these areas, cold air may collect or become stagnant. Least prone to low temperature and frost are the upper parts of steep slopes, where air movements lead to higher air temperatures close to the ground. The distance to large bodies of water, as well as the type of soil involved, also moderate this temperature pattern, however. Caution should also be taken in interpreting results for areas outside Asa Experimental Forest, where data for construction of the model were collected. In addition, the model is constructed for a heterogeneous forest cover and is not representative of conditions after extensive clear felling. Although the methodology does not provide quantitative assessments of the risk of frost damage, the risk classification obtained corresponds well with the distribution of observed frost damage among spruce seedlings within the Experimental Forest (Langvall et al., accepted; Langvall, 2000). The response and the frequency of frost would need to be incorporated into the methodology, however, in order for it to be able to provide a quantitative risk assessment of frost damage.

6.3.4 Acknowledgements

Several persons have contributed to this wind risk work: Dag Åstrand and Magnus Mossberg, IT Unit, Swedish University of Agricultural Sciences (SLU) are acknowledged for programming assistance, Dr. H. Peltola and Professor S. Kellomäki, Joensuu University, Finland, for making HWIND available to us and for modifying it according to our needs, Dr. B. Gardiner, Forestry Service, UK, for giving us access to the GALES model and Prof. G. Örlander (SLU) for providing windthrow observation data for Asa Experimental Forest, Sweden.

Chapter 7

Biogeochemical processes and mechanisms

Harald Sverdrup, Anna Hagen-Thorn, Johan Holmqvist, Patrik Wallman, Per Warfvinge, Charlotta Walse, Mattias Alveteg

7.1 Introduction

The modelling adapted in the SUFOR programme attempts to take a large step forward. This chapter describes some of the biogeochemistry processes needed in the next step biogeochemical modelling taken in SUFOR. There are several models for nutrient dynamics and growth available, however, development of these have in some respects moved in old tracks for the last decade. We see several drawbacks in these older constructs with respect to operationality and several new steps are necessary, especially with respect to nitrogen and carbon cycling, but also with respect to developing models that operate on observable parameters. What is described in the following is the result of several theme groups that have been operative in the SUFOR Programme. The groups have been multidisciplinary, which was a great help for innovation.

For tree health and vitality, and for predicting the effect of air pollution on forest health, the research climate was largely hostile towards any modelling attempt, despite the evident shortcomings and failures of the existing linear empiricism. A priori proof was demanded for every risk assessment or scenario, often in open contempt of the precautionary principle. During the years of acid rain research, large interest and effort was directed towards prediction of air pollution effects on forest ecosystems. Significant progress was also made by internationally and multidisciplinary researcher teams, sometimes overtaking the traditional forestry research efforts on their home ground.

The time period from 1988 to 1999 brought large changes in pollutant climate. European negotiations under the UN/ECE LRTAP convention focused on critical loads for deposition of sulphur and nitrogen brought models into the negotiating room. Not only predictions for critical loads and exceedances, but predictions of effects and environmental benefits of emission reductions was demanded. With the demand for predictions, modelling based on robust interpretation of experimental data become a necessity, overruling the earlier scientific conservatism and lack of system perspectives. New actors entered the modelling scene and performed the desired tasks. We have developed new models for the nitrogen and carbon cycle within SUFOR, trying to step one step back and restart our model formulations on a firmer mechanistic basis. These go back to discoveries for several of the components made several years ago, then they were too new and too much conflicting with the prevailing paradigm, and because of this they could not be published then. Since then, times have changed and the acceptance and understanding of modelling has grown significantly. What was started then reflect what we need now and what the data then and now tell us we should be doing. In the following, a review is

allocation patterns, nutrient re-translocation within the plant or changes in root exudation, but that could be accounted for later for example by putting different coefficients in the model. Plant growth and consequently the plant biomass is dependent on nutrient concentrations within the plant in accordance with Liebig's law of minimum. Thus root and shoot growth (1) and (2) is a function of nutrient element (amount or concentration) in the plant. Different nutrients are needed in special proportions, which were shown to be quite similar for different tree species in laboratory experiments with small seedlings (physiological requirements) but could be different for trees growing in the field (ecological amplitude). Tree species differ in the absolute amounts of nutrients needed to produce the same amount of biomass, the biggest difference being between deciduous and coniferous species.

There are different terms to describe biomass production in relation to nutrients in the plants. Nutrient use efficiency and nutrient productivity (Ingestad, 1979) are most often used. Nutrient use efficiency (as defined in Lambers et al, 1998) take into consideration mean residence time of the nutrient in the plant and therefore is better suiting for our purpose as we are not looking separately on nutrient losses from the plant. Nutrient use efficiency as well as nutrient productivity depends on plant genotype and environmental factors. Nutrient use efficiency at a whole plant level also includes re-translocation of nutrients from older leaves and roots to growing meristems, which occurs with nutrient shortage. As re-translocation is a typical reaction to nutrient deficiency at least for mobile nutrients, it can postpone growth reduction.

Thus leaf or needle lose in condition of severe nutrient deficiency can occur. That will reduce tree crown, amount of fixed carbon and tree growth. Until other environmental factors will limit growth more nutrients in the plant will produce more biomass even if plant also have to use nutrients for other purposes. Nutrients can also be stored for later usage and so called "luxury consumption" (Chapin, 1980) often occurs. Used nutrients have to be compensated by nutrient uptake. Nutrient uptake depends on plant nutrient demand (here element in the plant), nutrient absorption surface, efficiency of this surface in taking up nutrients and nutrient availability at the root surface. In case of nutrient deficiency in the plant all these components can be changed. As already mentioned, growth and nutrient demand could be decreased, it will happen quicker in response to deficiency of some elements than the others (see part II of my text) and thus some nutrient imbalances could occur. Nutrient absorptive surface can be enlarged by increased biomass allocation to the root, alteration of the root structure and, for mycorrhizal plants, increased degree of mycorrhizal colonisation. The balance between root or shoot growth and as the result root:shoot ratio is dependent on nutrient concentration in the plant. It is well known that root/shoot ratio is increased under condition of N, P and very likely S deficiency (Ericsson, 1995; Lambers et al., 2000 and references therein). In strictly controlled nutrient conditions the opposite tendency was observed in small birch plants deficient in Mg and Mn and also K (Ericsson, 1995). The author has mentioned, however, that differences between different species in the way potassium affects growth are likely to exist. In experiments with kohlrabi, for example, different reaction to potassium limitation was observed: K deficiency altered the source:sink relationship in favour of the roots (Singh and Blanke, 2000). Structure of the root system is a genetically defined plant characteristics, but it could be considerably modified by nutrient limitations as well as by root growing media (soil compaction, for example). Increase in number of root hairs and increased specific root length

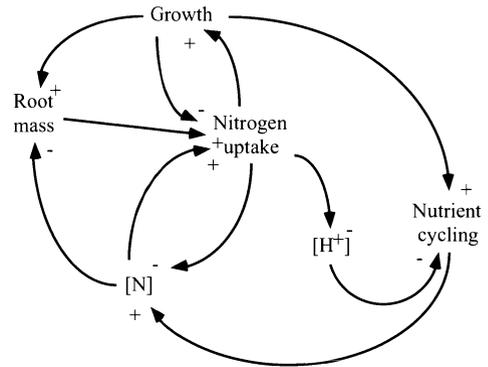


Figure 7.2: Causal loop diagram for the nitrogen uptake system of a tree.

are often observed in plants under limited nutrient supply. When amount of available nutrients at the root surface and the nutrient concentration in plant decrease root uptake kinetic parameters could be changed in a way to compensate this decrease (Marschner, 1995). Plant capacity to absorb element in shortage increases in response to deficiency of this nutrient due to increase in abundance and specific activity of transport proteins in the plasma membrane carriers. This change could happen very quickly: days and hours, which allow plants to regulate its nutrient uptake rapidly (Lambers et al., 1998). This response is found in many different species (Robinson, 1996). Plant ability to adjust root uptake kinetics parameters allow them to take up nutrients at extremely low concentrations, and according to Berendse et al., (1999) "in most species the value of K_m (the concentration in the solution where influx is equal to half of the maximal one) for N, P and K is so low that the concentration of these nutrients at the root surface can become virtually zero". That means that plant uptake capacity at least for these elements is very unlikely to be a uptake rate limiting step and nutrient transport to the root surface will define the rate of nutrient uptake. Plants are directly or indirectly influencing the availability of the nutrients in the root zone. The movements of nutrients to the root surface are dependant on plant water and nutrient uptake, and root nutrient interception depends on root growth into nutrient rich patches. Root exudation is strongly influencing rhizosphere chemistry, and composition and amount of root exudates is dependent on plant nutrient status. Exudation of organic acids, carbohydrates and amino acids stimulate microbial activity. In some cases, depending on competition for nutrients with microorganisms, it can provide more readily available nutrients at the root surface (Zak et al., 1994). There are also indications that P and Fe deficient plants can exude more organic acids in order to increase P availability in the rhizosphere, but this far it was shown to be important only for plants growing in calcareous soils. (Ström, 1998). In some forest species organic acids can neutralise Al, which inhibits plant nutrient uptake in acidic soils (Schöttelndreier, 2001). It is not likely, however, that this mechanism is of any considerable importance for forest trees. Heim et al., 2000 have concluded that at least for Norway spruce, Al tolerance could be rather attributed to immobilisation in the root apoplast than complexation of Al by root exudates. Most of common Swedish forest species found in the forests are colonized by ectomycorrhizal fungus. Hyphal mantles around the fine root tips could block the nutrient uptake by root itself (Smith and Read, 1997). Mycorrhiza hyphae take function of root hairs. They have smaller diameters and greater surface/volume ratio than plants roots or root hairs and can the nutrient absorptive surface can increase enormously. That's why mycorrhizal symbiosis is most important for acquisition of immobile nutrients, especially phosphorus, though mycorrhiza can be involved in plant uptake of several other elements (George and Marschner, 1995; Smith and Read, 1997). Ecto-mycorrhiza can take up organic nutrients (N, P) (Marschner and Dell, 1994) and like plants root it can produce exudates. In return for received nutrients plant is supplying assimilates for mycorrhizal growth and respiration, which is a cost for plant. That means that for plants growing in good soil condition, the mycorrhizal symbiosis will be than less profitable than for plants growing in poor soils. It could also have different uptake capacity per surface compare to plant root). It is not totally clear in which extent forest trees can regulate their mycorrhizal infection and once established to get rid of. A degree of mycorrhizal infection is dependant on root growth as it takes some time for mycorrhiza to infect and cover the root. Big long roots of forest trees are left non-mycorrhizal, but probably they do not contribute this much to tree nutrient uptake compare to fine roots. The reasonable way of modelling nutrient uptake of forest trees growing in the forest is to look at roots and its mycorrhiza as a whole without distinguishing between those two. The scope of the CLD discussed, do not include several other question that could influence nutrient availability at the root surface which related to exchange between rhizosphere and bulk soil and between different nutrient pools in the soil and. Spatial soil nutrient variability is also not considered. Localised nutrient supply may, in reality, influence structure and physiology of tree root systems at least in condition of nutrient deficiency, but the response could be different for different species, and at least for coniferous trees it was less pronounced (George et al, 1997) than for non-tree species (Robinson, 1996).

7.2.2 Nitrogen tree uptake processes

Several sources will supply N to the soil under present natural conditions in Northern Europe. The most significant quantitative input to natural ecosystems is today from the atmosphere as deposition of NH_4^+ and NO_3^- . Under natural unpolluted conditions, the N cycle will be in or close to balance, giving rise to no large net fluxes of acidity or alkalinity in the ecosystem. This is however changed if large sources such as deposition and fertilization or sinks such as enhanced forest growth are created in the system. The decay of litter and organic material will also be able to produce NH_4^+ ions. A part or all of the available N will be taken up by the root systems in the soil and simultaneously undergo nitrification to NO_3^- . Uptake of NH_4^+ is generally caused by adsorption/ion exchange mechanisms at all the root cell membranes, and the influx of NH_4^+ ions are balanced by the outflux of another cation, H^+ or potassium. When NO_3^- is taken up, this is through similar processes for anions, where the charge balance is kept by the excretion of another anion, HCO_3^- or OH^- . NO_3^- is also thought to be taken up passively by mass flow of water through root tips, and that this occur at a rate per surface area approximately ten times faster than the ion exchange mechanism acting on all root surfaces. For most plants the uptake of NH_4^+ is energetically more favourable than uptake of NO_3^- . Taking up NO_3^- is less advantageous as compared to NH_4^+ , because the plant must first convert the NO_3^- to NH_4^+ . The preference for NH_4^+ would be dependent on the chemical energy potential, and accordingly also depend on the relative soil solution ratio of NH_4^+ to NO_3^- . This would be consistent with the empirical observation that N uptake appear in the database of the Swedish forest survey to be proportional to the total N soil solution concentration. N uptake increase along the north to south axis in Sweden is stronger than the temperature dependence of uptake of base cation would indicate, indicating that the uptake of N is also driven by the N concentration. On the average the net uptake is in the range of $0.1\text{--}0.2 \text{ keq N ha}^{-1}\text{yr}^{-1}$ in the far north where the average temperature is in the range of -1 to $+1$ °C, and the precipitation 500mm. In the south the uptake increase to $1.4 \text{ keq ha}^{-1}\text{yr}^{-1}$, the annual average temperature is 8°C and the precipitation. If the uptake has the same temperature dependence as the uptake of base cations, then

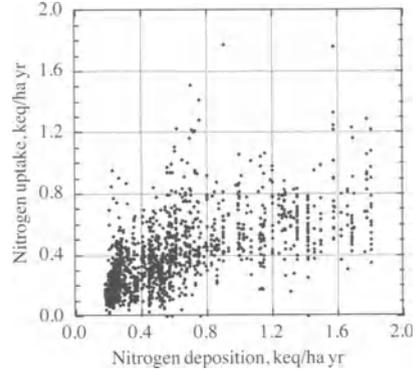


Figure 7.3: The relation between nitrogen deposition and forest net uptake in Sweden, using the Forest Survey data. The data show a distinct relationship between N deposition and growth. The effect of temperature has been removed. Data from 1884 sites were used.

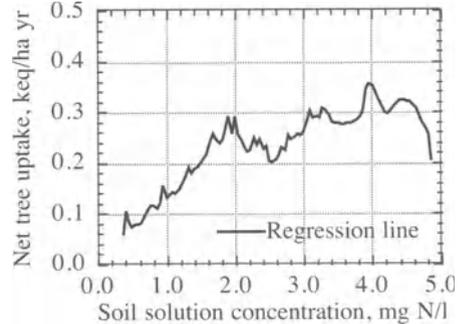


Figure 7.4: The relation between reconstructed average soil solution N and net uptake of total nitrogen in Swedish forest soils, using the Forest Inventory data. The line represents the running average. It can be seen that the curve show typical Michaelis-Menten behaviour, with a first order part and a zero order part at high concentrations. The effect of temperature has been removed from the data before plotting.

the temperature dependence from north to south would drive the uptake of $0.20 \text{ keq ha}^{-1}\text{yr}^{-1}$ up to $0.40 \text{ keq ha}^{-1}\text{yr}^{-1}$. The difference would be the uptake promoted by increased nitrogen deposition, and suggest a first order relation with respect to total N concentration.

Uptake kinetics

The uptake kinetics for a tree population on a per average tree basis appear to follow (Sverdrup, Warfvinge and Rosen, 1991):

$$r_{\text{uptake}} = k_N \cdot \frac{[N]}{K_N + [N]} \cdot \prod_i f_i \quad (7.1)$$

where f_i is different modifying functions taking account of the effect of soil chemistry (Al, (Ca+Mg+K), pH, heavy metals) and climatic factors (Soil moisture, temperature). A very simplified version of this equation would be;

$$r_{\text{uptake}} = k_N^* \cdot \frac{N_{\text{dep}}}{K_N^* + N_{\text{dep}}} \cdot \prod_i f_i \quad (7.2)$$

assuming that soil concentration under a high nitrogen pollution load become to a large degree controlled by the amount of deposition. From this we have that when:

$$K_N \gg [N] \quad (7.3)$$

then:

$$k_N \cdot \frac{[N]}{K_N + [N]} \rightarrow \frac{k_N}{K_N} \cdot [N] \text{ and } r_{\text{uptake}} = \frac{k_N}{K_N} \cdot [N] \quad (7.4)$$

and when:

$$K_N \ll [N] \quad (7.5)$$

then:

$$k_N \cdot \frac{[N]}{K_N + [N]} \rightarrow k_N \text{ and } r_{\text{uptake}} = k_N \quad (7.6)$$

In order to parameterize the expression, we will use the soil database available within the Swedish Forest Inventory (NFI). This is a continuous survey of key parameters at 27,000 locations in the Swedish forest, in order to monitor forest state, vitality and production capability. Of these, weathering rates for Ca, Mg and K are available at 1884 locations. In the soil survey uptake figures for N is available. Swedish forest ecosystems appear not to decompose litter at the same rate as litterfall is generated at present. This may however be a transient phenomenon, leading to increased leaching in the future. The relation in Fig. 7.4 and N deposition can be used to calculate the concentration in the rooting zone of the soil. N uptake to plants in Sweden is shown in Fig. 7.6 along with the observed deposition, and if this is converted to concentration, as in Fig. 7.6 versus the calculated soil concentration for total N. The average soil concentration was calculated as:

$$[N] = \frac{D}{2 \cdot P} + \frac{D - U - Im}{2 \cdot Q} \quad (7.7)$$

The correlation between the parameters is evident. The values have been filtered in Fig. 7.4, and a running average has been fitted to the data. Then the typical Michaelis-Menten curve appears. This allow us to parameterize the uptake rate equation from Fig. 7.3 and 7.4:

| Driving force | k_N | K_N |
|------------------------------|---|--|
| Deposition based kinetics | 0.8 keq ha ⁻¹ yr ⁻¹ | 1.23 keq ha ⁻¹ yr ⁻¹ |
| Concentration based kinetics | $0.3 \cdot 10^{-3}$ keq ha ⁻¹ yr ⁻¹ | $1.5 \cdot 10^{-3}$ kmol m ⁻³ |

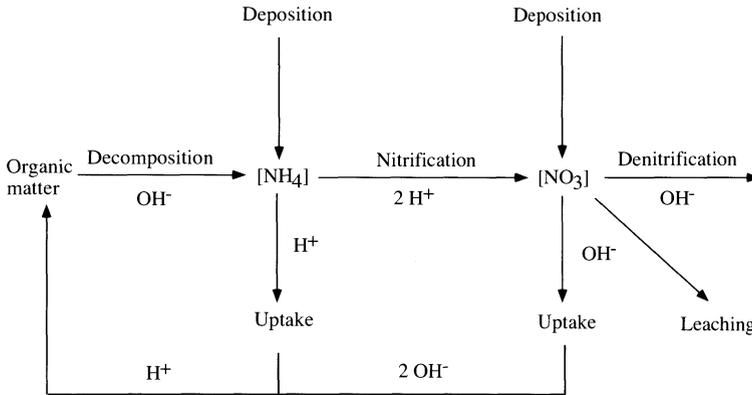


Figure 7.5: The effect of nitrogen uptake, nitrification, denitrification or leaching on the ANC balance in the soil.

Retarding factors have been applied to several types of processes, and are sometimes used in industrial reaction engineering science (Sverdrup et al., 1984, 1991; Levenspiel, 1968). Here the retardation function is composed of different elements:

$$\prod_i f_i = f(BC/Al) \cdot f(\theta) \cdot f(T) \quad (7.8)$$

where $f(BC/Al)$ is the function taking the effect of pH, Al and Ca into account, $f(\theta)$ the effect of soil moisture and $f(T)$ the effect of temperature. The soil moisture function for N uptake would probably be similar to the function for base cation uptake (Sverdrup, Warfvinge and Rosen, 1992), the acidity-heavy metal function has not yet been determined, but is known for base cation uptake. In general the limiting nutrient is the overall limiting growth factor, and the tree will be limited by either the N, P or base cation (BC) supply. This allows us to write the general expression:

$$U_{crit} = \min\left(\frac{U_{critN}}{x_N}, \frac{U_{critCa}}{x_{Ca}}, \frac{U_{critMg}}{x_{Mg}}, \frac{U_{critK}}{x_K}, \frac{U_{critP}}{x_P}\right) \quad (7.9)$$

x_i is the stoichiometric ratios for phosphorous, nitrogen and base cations in the tree. This is complicated by the fact that the tree will not utilize more nitrogen for building biomass than the supply of other nutrients can permit, but uptake may still be larger, and the excess excreted over the leaves and leached by rain to the ground. For most forests in Sweden so far, the uptake and forest growth has been limited by the nitrogen supply. The uptake of NH_4^+ and NO_3^- can be considered as being separate mechanisms and both dependent on substrate concentration. NH_4^+ is a monovalent ion and would compete with H^+ -ions and K^+ -ions at the root surface when they are present in high concentrations. NO_3^- is also monovalent negative and would adsorb mainly in competition with OH^- and HCO_3^- , while Cl^- is poorly adsorbed in soils, depending on selectivity and concentrations. Presuming normal ion exchange behaviour, the surface concentration would be linearly proportional to the soil solution concentration. This is an assumption however, and needs experimental verification. Under extreme depositions of NH_4^+ or NO_3^- the additional N does not promote further increase in growth, indicating that there is some threshold above which the uptake becomes concentration independent. This is the typical behaviour of Michaelis-Menten kinetics, often seen in kinetics of microbiological processes:

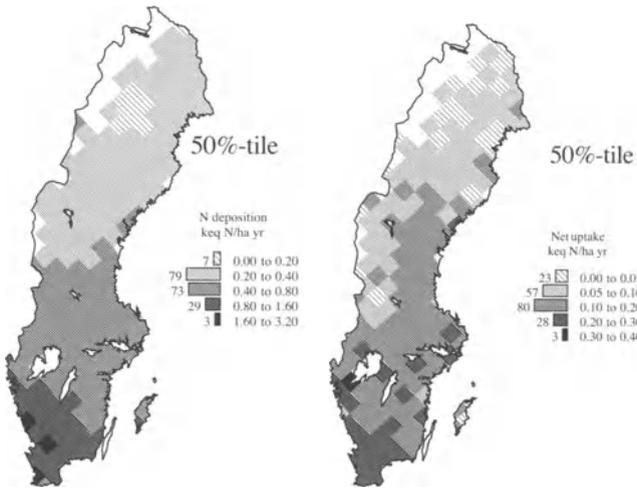


Figure 7.6: Total N deposition to Swedish forest related to tree uptake to stem biomass. 1.0 keq N corresponds to 14 kg. nitrogen. 0.2 keq/ha yr 2.8 kg ha⁻¹yr⁻¹, 0.4 keq ha⁻¹yr⁻¹ is 5.6 kg ha⁻¹yr⁻¹, 0.8 keq ha⁻¹yr⁻¹ is 11.2 kg ha⁻¹yr⁻¹, 1.6 is 22.4 kg ha⁻¹yr⁻¹. The data on net uptake projected to become removed in stem harvest in the Swedish forest. The data come the Swedish Forest Inventory, based on actual measurement of tree growth.

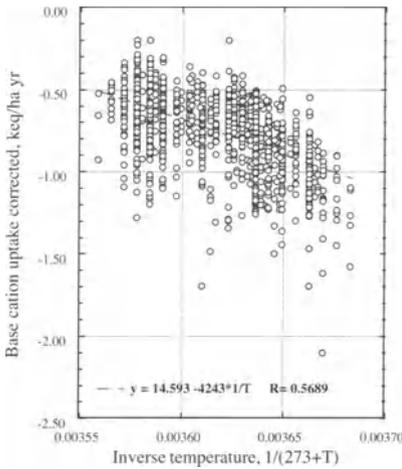


Figure 7.7: Data for forest growth in Sweden, used to determine the temperature dependence of growth, after removal of N-deposition promotion effects. The north-south gradient is not only caused by temperature, but also nitrogen availability.

$$r_{u,NH_4^+} = \frac{k_{NH_4^+} \cdot [NH_4^+]}{K_1 + [NH_4^+]} \cdot \prod_k f_k \quad (7.10)$$

where f represents the damage functions to the process. The corresponding expression for nitrate uptake is:

$$r_{u,NO_3^-} = \frac{k_{NO_3^-} \cdot [NO_3^-]}{K_2 + [NO_3^-]} \cdot \prod_m f_m \quad (7.11)$$

The uptake preference for NH₄⁺ would then be only partial, and in most soils only occurring in the top layer of the soil where the NH₄⁺ concentration is high. The damage functions depending on BC(Al) ratios have been determined earlier (Sverdrup et al., 1992) and data for the dependence on pH is shown in Fig. 7.8. The long-term uptake of N is defined as the N uptake that can be balanced can be balanced by a long-term supply of base cations. This amount is referred to as the critical uptake N_{crit} . The calculations are thus not based on the present uptake rate, since the present growth may be enhanced by artificial or time-limited supply of nutrients, mobilized from exchange sites by acid deposition. The critical uptake is calculated from mass balances for the nutrient cations Mg, K and

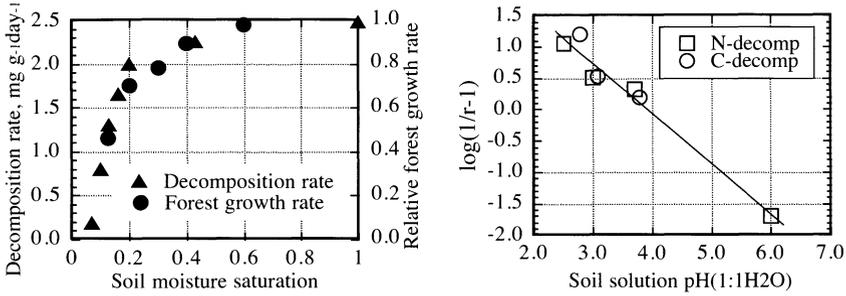


Figure 7.8: The dependency of N uptake to trees and organic matter decomposition in the soil as a function of the moisture saturation degree in the soil (Sverdrup et al., 1992). The dependence of the decomposition of organic matter expressed as $\log(1/r-1)$ and tree growth in forest soils on soil solution pH(H_2O). The logarithmic expression was derived from the kinetics and used to obtain a linear relationship to extract the water dependency coefficient.

| Tree species | Ca/N | Mg/N | K/N | P/N | BC/N |
|----------------|------|------|------|------|------|
| Norway spruce | 0.4 | 0.13 | 0.17 | 0.08 | 0.7 |
| Scots pine | 0.4 | 0.10 | 0.15 | 0.08 | 0.65 |
| European beech | 0.4 | 0.2 | 0.2 | 0.08 | 0.8 |

Table 7.1: Approximate nutrient ratios, based on uptake in $eq/ha\ yr$, to be applied in calculations to prevent long term nutrient imbalances. The values have been determined by studying the distribution of nutrient ratios at 1884 sites in the Swedish Forest Inventory soil survey program, as well as nutrient deficiency studies from the literature.

Ca separately. If weathering as separate ions are not available, then the total base cation weathering and a BC/N ratio can be used. The production of different cations from weathering can be calculated with the PROFILE model, from total analysis or estimated from soil type, parent material, temperature, texture and soil wetness. BC weathering rates are available in the SMB input data files. Taking deposition and weathering as the sources of these nutrients, and uptake and leaching as the sinks the mass balance for a species i becomes:

$$U_{crit,c} = D_c + W_c - Q \cdot [c]_{lim} \tag{7.12}$$

- where $U_{crit,c}$ = critical uptake of base cation c
- D_c = atmospheric deposition of base cation c
- W_c = production of base cation c from weathering
- Q = water flux from the bottom of the rooting zone
- $[c]_{lim}$ = limiting concentration for uptake of nutrient c

The limiting concentration for uptake of nutrient c is the level when the trees no longer can extract a nutrient from the solution. In the calculations, the limiting concentration for Ca and Mg has been set to $5\ meq\ m^{-3}$, and to 0 for K. From the critical base cation uptake, the corresponding N uptake can be calculated, by introducing the ratio between each cation and N in the biomass:

$$N_{u,crit} = \min\left(\frac{U_{crit,Ca}}{x_{Ca/N}}, \frac{U_{crit,K}}{x_{K/N}}, \frac{U_{crit,Mg}}{x_{Mg/N}}, \frac{U_{crit,P}}{x_{P/N}}, U_{N,present}\right) \tag{7.13}$$

where $N_{u,crit}$ = critical N uptake
 $x_{i/N}$ = ratio of nutrient i to N during uptake

It is assumed that uptake cannot be larger than at present, if present is set by standard yield tables or actual field measurements. A series of conditions must be obeyed for this approach to work. Uptake and immobilization due to retarded decomposition of litterfall cannot be larger than deposition. If there is no more N left after uptake and retarded decomposition immobilization, then both microbial immobilization and denitrification is set to zero. Both terms require concentration of N, but have lower priority than uptake by trees. At steady-state the, and with a balanced nutrient supply, the N leaching should amount to the natural leaching from N-limited stands. In the calculations performed for Sweden, initially the basic assumption behind the critical uptake concept was $N_l=0$. However, not all ecosystems are necessarily dominated by trees.

Uptake rate modifiers

The relation between forest yield and soil moisture classification has been derived earlier (Sverdrup et al. 1992), and is nearly identical to the relation observed for biomass decomposition in the upper soil layers, having the appearance of a Freundlich adsorption isotherm:

$$f(\theta) = \frac{k_w \cdot (\theta/\theta_m)^{n_w}}{1 + k_w \cdot (\theta/\theta_m)^{n_w}} \quad (7.14)$$

θ is the soil water content and θ_m is the maximum content, k_w is the value of the Freundlich adsorption isotherm water adsorption coefficient and n_w is the Freundlich adsorption isotherm adsorption order. The data are shown in Figure 7.8. $k_{water} = 110$ and $n_{water} = 2.4$. The appearance of the Freundlich adsorption expression makes it tempting to interpret the soil moisture saturation dependency as a result of a water adsorption to the tree roots and organic material in the soil profile. The near identical function for biomass decomposition and forest yield (presumably closely correlated to nutrient uptake), indicates that this may have to do with the basic mechanism for mechanical wetting of the soil constituents. The temperature dependence of growth is following an Arrhenius relationship:

$$r_{growth} = r_0 \cdot 10^{A/(273+T)-A/281} \quad (7.15)$$

Data from the literature (Sverdrup et al., 1992) indicate that $A=5200-4800$. At temperatures below 2°C , growth is assumed to come to complete halt. In Fig. 7.7, the temperature dependence as observed in the field data from the Swedish Forest Survey is shown. The uptake rates were normalized to the average soil solution concentration for all Swedish forest soils, in order to remove the deposition promotion effect overlying the temperature effect. The conclusion is less certain than the data from laboratory experiments, but Fig. 7.7 represents 1884 field estimates of growth from the Swedish Forest Inventory. In the field data, the growth appear to have an Arrhenius factor of approximately $A=4240 \pm 900$. The data may have been slightly underbiased by the normalization and the use of the base cation data, indicating that the real field value lie between the value derived from literature data on tree growth $A=3100$ and the value obtained here for base cation uptake, $A=4240$. The N soil solution concentration along the country is considerable stronger than the base cation gradient, and correcting for this and separating out brown soils with deciduous trees, is leading to a value of approximately $A=3700$ for nitrogen uptake. This indicate that the value for deciduous tree may be somewhat higher, $A=3900-4100$. The obtained values are significantly less than the value observed for organic matter decomposition in the soil, $A=5200$.

7.2.3 Base cation uptake processes

Uptake of any nutrients to plants is a multi-stage processes. Several different types of processes are involved, and the slowest step in the process chain will determine the total uptake rate. The

rate limiting step may differ for the same plant, depending on the conditions. For plant growth *Liebig's law* must be obeyed; The nutrient in least supply will determine the maximum growth rate G_{max} . Growth of plants and production of biomass is directly proportional to nutrient uptake, since different elements are incorporated in biomass in relatively fixed concentrations and proportions.

Under normal undisturbed or unpolluted ecosystem conditions, plant growth is mainly determined by water, nitrogen, temperature or light availability. In terms of the uptake process flowchart in Fig. 7.10, it can be seen that water availability influences soil solution concentrations and mass flow as well as sap flow in the plant. Under rare optimal light, water and temperature conditions, growth may consume nutrients to the degree where diffusion in the root vicinity can become rate limiting. Root ion exchange of nutrients is normally not rate limiting in normal soil conditions, with little anthropogenic acidification. For most natural European forests, the normal condition is that the forest should be nitrogen limited. Then N availability in the soil solution and the solute flow of N will be the growth rate limiting step. Historically, N input to European forest ecosystems were 1/5 to 1/10 of the present 1990 input. Base cations under most conditions are available in sufficient supply. However, under soil acidification conditions, other positively charged ions besides Ca, Mg and K start to appear in the soil solution in high concentrations (Al, H, Fe), and the ratio of these ions to base cations and P is greatly changed. Fig. 7.9 show the causal loop diagram for Calcium and Magnesium uptake to a plant and the feedbacks connected to that process. The changed solution concentrations and ion ratios will result in a different surface composition of adsorbed ions on the root surfaces. The transport mechanisms taking ions through the root membrane into the sap in the root, depend on the availability of necessary nutrient ions on the outer surface of the root membranes. With uptake of base cations like Ca or Mg severely restricted due to low root surface concentration, uptake of base cations and phosphorus may become growth limiting instead of nitrogen. This leads to a number of statements concerning the mechanisms of the uptake process:

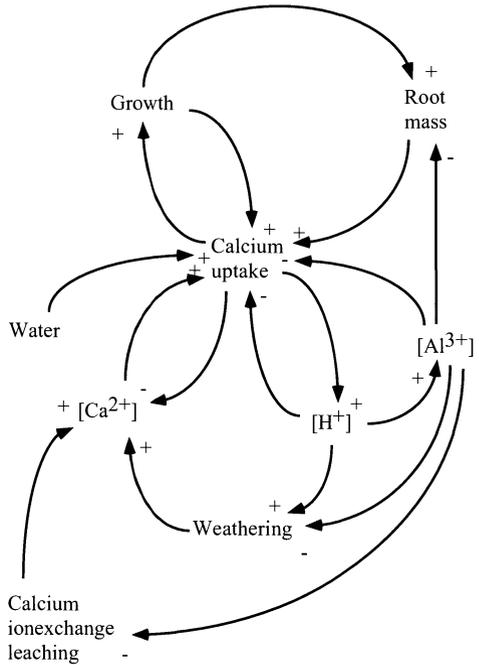


Figure 7.9: Causal loop diagram for the base cation uptake system of a tree.

- Base cation uptake occur in several steps
 - Solution transport to the root
 - Diffusion in the rhizosphere
 - Ion exchange to bind at the surface
 - Active uptake through the root membrane
- Al^{3+} , Ca^{2+} and Mg^{2+} compete for the same sites at the root
- H^+ , NH_4^+ and K^+ compete for the same sites at the root

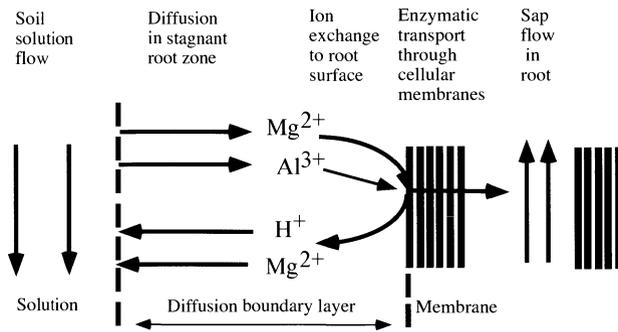


Figure 7.10: Uptake of nutrients necessary for growth of plants occur through a series of processes, each of which may be rate limiting, depending on the conditions in the soil and the geometry of the root boundary layer.

- The uptake capacity is finite
- The transport rate through the membrane is dependent on the concentration at the root membrane surface
- The role of water is to set the activity of the root surface

This implies that the uptake process is considered to be independent of the water uptake and water flux into the root. Fig. 7.10 show a conceptual model for tree growth a process. According to this view, maximum growth is ultimately limited by plant physiology. The first regulator of growth is light, since this is significant for total energy available to the plant. Temperature will modify the chemical processes involved in photosynthesis and energy conversion. Water will affect the efficiency of the root apparatus as well as transport of substance internally in the plant. These factors will modify physiological maximum growth to a maximum possible growth given the light, temperature and water conditions. Availability of nutrients as applied in "Liebig's law" regulates potential growth to real growth, ignoring environmental retarding effects. Under acidification, soil aluminium and acidity may further limit growth.

Earlier model approaches considered

Earlier models for tree growth (Ågren, 1983; Aber et al., 1991; van Dam, 1992; Nye and Tinker, 1977) have tended to model growth as a function of mass flow and availability of nitrogen, water and light. This concept will work well in forest stands only affected by such factors, untouched by pollution, climate change or large soil chemistry changes over time. The light/nitrogen limitation concept has however failed to predict or explain forest decline caused by soil acidification, due to the fact that important processes and couplings to soil chemistry are missing in these earlier models. Incorporating such connections lead to predictive capability concerning soil acidification effects (Bossel et al., 1985). The concept of Nye and Tinker (1977) assumes ion exchange at the root surface to be the step preceding uptake. The next step is uptake of adsorbed ions from the root surface according to first order or Michaelis-Menten kinetics:

$$r_{up} = k_{up} \cdot \frac{X_{BC}}{X_{BC} + K_M} \quad (7.16)$$

This expression include the base saturation explicitly, but left the capacity CEC to be non-explicitly included in the constant. Uptake from the outer root surface through the cell wall

into the plant is actively regulated by the plant. But transport from the bulk of the liquid to the outer root surface is passive with respect to the plant and governed by physical processes such as flow, convection and diffusion. Combining this with surface concentration control by ion exchange as outlined earlier, or by a simple Langmuir ($z=1$) or Freundlich (any z) adsorption isotherm for base cation adsorption:

$$Ads_{BC} = Ads_{max} \cdot \frac{[BC]^z}{[BC]^z + K_{adsBC}} \quad (7.17)$$

We can derive an uptake expression, used by several authors earlier (Nye and Tinker, 1977; Gherini et al., 1990; van Oene, 1993), by inserting in the Michaelis-Menten expression. This traditional view of uptake does not envision any drastic variation in Al in the soil, nor that there may be interactions between ions at adsorption sites on the root. It cannot predict what would happen if Al were to change significantly, and it cannot predict what increased Al in soils would do to tree growth. This implies that these traditional uptake models cannot predict any change in growth due to soil acidification. They simply assume that soil acidification does not occur. In unpolluted soils, there will be much Al in the exchange complex, but very little in solution. When there is a significant increase in soil solution Al, then the basic assumptions concerning BC uptake are no longer valid. Under such conditions, adsorption of Al to the root will increase and this will change the amount of adsorbed BC at a certain soil solution concentration. The amount of adsorbed BC must share space on the root with Al, the fraction of adsorbed BC:

$$X_{BC} = \frac{Ads_{BC}}{Ads_{BC} + Ads_{Al}} \quad (7.18)$$

This can be inserted in the Michaelis-Menten expression for uptake traditionally used:

$$r_{up} = \left(\frac{k_{up}}{1 + K_M} \right) \cdot \left(\frac{[BC^{2+}]}{[BC^{2+}] + [Al^{3+}] \cdot \left(\frac{K_M}{1 + K_M} \right) \cdot \left(\frac{[BC^{2+}] + K_{adsBC}}{[Al^{3+}] + K_{adsAl}} \right)} \right) \quad (7.19)$$

K_M is the Michaelis-Menten half-rate-saturation coefficient. The full Michaelis-Menten expression of Nye and Tinker (1977) may possibly not be necessary, the first order approximation of it may suffice, because uptake will level off when the surface is saturated with BC. At this point it is not reasonable to continue with purely empirical and linear only adsorption isotherms, when ion exchange expressions derived from the law of mass action are available, also accounting for non-linearities (Vanselow, Gaines-Thomas, Gapon e.t.c.).

Model assumptions

It is assumed that base cation uptake can become uptake rate limited in severely acidified soils, and soils receiving high nitrogen deposition. It is assumed that this can be described by a model where uptake is dependent on absorbed amount of the ions taken up on the root. The surface ion exchange effect can be expressed to a large part by using the soil solution ratio between the divalent base cations Ca, Mg, K and Al. (This is often referred to as the "Ca:Al-ratio" in the literature, even if it is understood that it should comprise Ca, Mg and K, as "Al" should rather be understood as the sum of all inorganic charged Al-species (Bonneau, 1990)). In the continuation we will sometimes use the notation $BC=Ca+Mg+K$. It becomes increasingly more difficult for the plant to take up the necessary base cations from the soil solution when soil acidification occur, due to the increased concentration of H^+ and Al^{3+} in relation to base cations. The following assumptions form the basis of the $(Ca+Mg+K/Al)$ -response functions extracted from the experimental data:

- Uptake of base cations is proportional to the adsorbed amount of Ca, Mg and K on the root surfaces active in uptake.

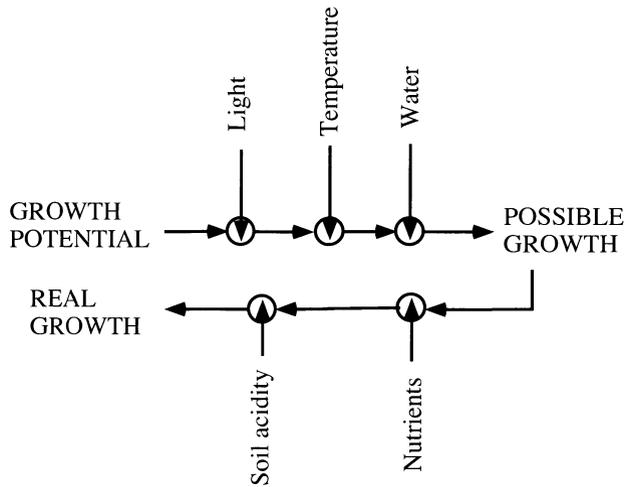


Figure 7.11: *Conceptual model for tree growth as a process. Growth is ultimately limited by plant physiology. The first regulator of growth is light, since this is significant for total energy available to the plant. Temperature will modify the chemical processes involved in photosynthesis and energy conversion. Water will affect the efficiency of the root apparatus as well as transport of substance internally in the plant. These factors will modify physiological maximum growth to a maximum possible growth. Availability of nutrients as applied in "Liebig's law" regulates potential growth to real growth. Unfavourable soil chemistry may further limit growth.*

- Al may disturb the uptake mechanisms at the root surface, by competition at adsorption sites, in the root membrane and cytoplasm.
- The concentration at the root surface is controlled by an adsorption mechanism, analogous to ion exchange between the major soil solution constituents Al^{3+} , Ca^{2+} , Mg^{2+} , K^+ and H^+ ions.

The ion exchange analogy is proposed with special reference to Asher, (1987); Bolt (1982); Cronan, (1991) and Morris et al., (1989) but also with respect to earlier studies (Sverdrup et al., 1990, 1992) and results presented by Cronan et al., (1989); Cutler and Rains, (1974); Eriksson (1988, 1989); Goyer, (1988); Hüttermann and Ulrich (1984) and Ulrich, (1983, 1984).

- Growth is reduced if the possible uptake of BC is lower than what is required to physiologically match the potentially available uptake of N (i.e. Liebig's law applies to trees and BC and N uptake).

In addition to the causes given above, high soil solution concentrations may cause denaturalization of extracellular and intracellular enzymes and by immobilization of P in the soil. This is not incorporated in the assumed mechanisms and the kinetics derived thereof. The value of the $(\text{Ca}+\text{Mg}+\text{K})/\text{Al}$ -ratio in the soil solution is coupled to uptake of base cations, chemical weathering, base cation deposition and acid deposition. Uptake will tend to make $\text{Ca}+\text{Mg}+\text{K}$ smaller in the soil, weathering and atmospheric deposition of $\text{Ca}+\text{Mg}+\text{K}$ will tend to increase it, whereas acidification will tend to increase Al. Thus the net effect of forest growth and acidification may be similar with respect to the $(\text{Ca}+\text{Mg}+\text{K})/\text{Al}$ -ratio, even if the damage mechanisms are very different on the molecular level at the root-soil solution interface. In the short term

perspective, (Ca+Mg+K)/Al ratios and soil acidification aspects are of interest for forest stand health and vitality, in the long term for forest productivity and hence commercial value.

Uptake kinetics

The uptake of Ca, Mg and K here represented as BC, is governed by a kinetic equation based on soil solution concentration of base cations, usually a Michaelis-Menten expression. When base cations are limited in supply, but under conditions favourable for uptake, then uptake will be concentration dependent, approaching zero order when base cations are available in large supply. When growth is restricted by some other nutrient, uptake will also be zero order. However under acidic soil conditions the availability of base cations may become restricted by supply or by the chemical conditions in the soil which would allow other constituents of the soil solution to compete for root receptor sites. Uptake U is expressed as a function of a plant physiologically determined maximum uptake, modified by soil moisture and soil chemistry functions. Growth is proportional to nutrient uptake, limited according to "Liebig's law" and regulation by climatic factors such as light, temperature and soil water availability:

$$G_{min} = \frac{U_{limit}}{x_{limit}} \cdot f(light) \cdot f(\theta) \cdot f(T) \tag{7.20}$$

$f(\theta)$ is the soil water availability function, which has been described elsewhere (Sverdrup et al., 1992). Modifiers taking into account the effect of temperature ($f(T)$) and light $f(light)$ have been omitted for simplicity in the following text. When N or P is present in large quantities, then this can temporarily affect the growth capacity of the plant, for U_{limit} , Liebig's law prevails, in the long run.

Under growth conditions restricted by soil acidification, uptake become dependent on the amount of base cation adsorbed at the tree root:

$$U = k \cdot BC_{ads} \tag{7.21}$$

where U is the uptake rate, k is the uptake rate coefficient. The amount of adsorbed base cation at the root is given by the adsorption capacity CEC_{root} , the available root surface and the base cation saturation at the root exchange sites:

$$BC_{ads} = A_{root} \cdot CEC_{root} \cdot X_{BC} \tag{7.22}$$

BC_{ads} is the adsorbed amount base cations, A_{root} is the total active surface at full wetting of the roots, X_{BC} is the fraction of the adsorption sites on the root occupied by Ca and Mg, CEC_{root} the adsorption capacity (keq/m² root) and U the uptake rate. The effective root surface is defined as the root surface exposed to water in the soil to the degree where the water is also in contact with other soil constituents supplying nutrients. The maximum uptake U_{max} as limited by the Michaelis-Menten expression only, is occurring at full base cation saturation at the root surface, when $X_{BC}=1.0$. U_{max} is basically dependent on the cation exchange

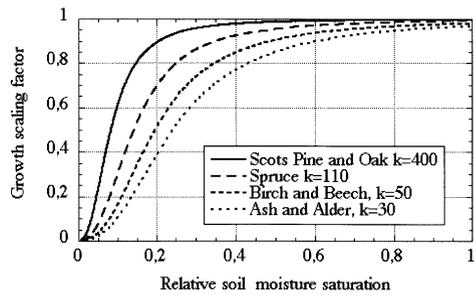


Figure 7.12: The soil moisture dependency of nutrient uptake to different types of trees. The different moisture isolines were derived by scaling from Norway spruce as a reference. The equation $f = k \cdot \theta^n / (1 + k \cdot \theta^n)$, where $n=2.4$ and the coefficient k vary with tree species; For drought-tolerant species, Scots pine and Oak, $k=400$, for intermediately sensitive, Norway spruce $k=110$, for drought intolerant trees, Beech and Birch, $k=50$ and for the trees very sensitive to drought, Ash and Alder, $k=30$.

capacity, specific active surface area and the transport rate through the membranes of the root. If the proper expressions are combined, the kinetic uptake equation is derived:

$$U = U_{max} \cdot X_{BC} \quad (7.23)$$

Changes in root mass over time as the tree grows imply a change in A_{root} over time. The non-specific divalent adsorption is assumed to occur analogously either to Gapon, Vanselow equilibrium ion exchange, or valence unspecific ion exchange. Uptake of Ca, Mg and Al and their soil solution seem to be strongly coupled, the uptake of K and NH_4 affect each other mutually, but are less sensitive to Al interference. Phosphorus as PO_4 and NO_3 are also taken up in a series of similar events, mainly as negative ions. In multi-layer assessments, it will be necessary to consider that the BC/Al value is not uniform down through the soil profile. This will allow the plant to partially reallocate uptake to other soil layers, if the conditions become adverse for uptake in a particular layer. However if the PROFILE model is used (Sverdrup and Warfvinge, 1988; Warfvinge and Sverdrup, 1992), uptake in any layer will be limited by actual supply. In a long term perspective, nature will limit this to the amount available from weathering in that particular layer plus any percolate in excess of $15\mu\text{eq/l}$. Under conditions where base cation supply is limiting growth, the limiting uptake will be:

$$U_{limit} = \min_{j=N,P,Ca,Mg,K} \sum_{i=1}^{layers} u_{crit,i}(j) \cdot f\left(\frac{BC}{Al}\right)_i \quad (7.24)$$

$u_{crit,i}$ is maximum uptake possible in layer i of nutrient j . The critical uptake is determined by availability of the nutrient in the soil. For N, deposition, fixation and decomposition of organic matter is the most important sources. For P and base cations, weathering and atmospheric deposition are most important in the long term, in a shorter perspective, ion exchange may be an important source. For considerations of long term availability, i_{ex} and i_{decomp} must be set to zero.

$$u_{crit,i} = \eta \cdot (i_w + i_{EX} + i_{DECOMP} + i_{DEP} - i_L) \quad (7.25)$$

where η is the root efficiency, i_w is release rate due to weathering of nutrient ($i=N, P, Ca, Mg, K$), i_{dep} is input of i from the atmosphere, i_L is the amount leached at the bottom of the root zone of nutrient i . The effect of the above description of uptake kinetics is that the tree will compensate for a lowering of BC/Al ratio in the B-layer, by increasing uptake from other layers. Fine root mass will be reallocated in the soil over longer time periods to soil layers with better BC/Al ratio as long as this is possible. When the amount available in a certain layer is not corresponding to the amount uptake reallocated to that layer by the plant, and no other layer can supply the missing amount, then growth is reduced. Thus the plant will try to optimize its uptake. When one investigator reports significant growth changes at 2.5 mg/l,

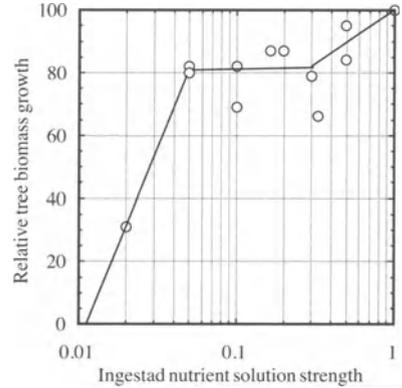


Figure 7.13: The dependence of growth on nutrient solution concentration be investigated by using data from Arovaara and Ilvesniemi (1990) and Ilvesniemi (1992). The data shows that plants can take up nutrients down to a certain limiting solution concentration. This shows that uptake depend on concentrations in the solution and not available flux as sometimes stated. It also shows that these concentrations may actually occur under special field conditions.

whereas another reports no change until 15 mg/l Al or more, then this difference can often be traced back to differences in Ca, Mg and K concentrations of the soil solution of the bioassay. More often than not, K concentrations will be several orders of magnitude larger than seen in nature. This is caused by the use of "Ingestad" or Clarke" or Hoagland" nutrient solutions or similar compositions, rather than something similar to the natural soil solution composition. This implies that K and Ca concentrations may be in the range of 100-20 mg/l, whereas in the soil, but more than 1-2 mg K/l and 2-6 mg Ca/l is seldom observed under field conditions. The use of abnormal K-levels in many experiments are a major confounding factor in the evaluation of these particular experiments. A best fit to the data in Fig. 7.13 would yield:

$$f(BC) = 0.9 \cdot \frac{[BC]^2}{([BC]^2 + 0.0007)} \quad (7.26)$$

when the concentration is expressed as multiples of Ingestad solution strength. In the Asa study, such a relationship between soil base saturation and content in needles of Ca, and Mg was found. If we assume that the base saturation of the soil is correlated to the base saturation of the roots, the relationship appear to be very probable. In Asa study, the root efficiency could also be determined by mass balancing in a number of stands.

7.2.4 Modelling forest growth

Modelling of forest growth is based on principles from the PnET and FORSANA models. Much of what follows describes the philosophy used in these models. Before any physiological calculations can be done in the PnET sub-model, the biomass content of the different tree compartments must be updated. The tree is considered consisting of seven compartments, foliage (divided into a user-defined number of crown-layers and up to six different age classes), branches, sapwood, coarse roots, fine roots (in each of a user-defined number of soil layers), regenerative tissue and freely available carbohydrates. There is also a compartment of heartwood. It does not have any influence on the physiological processes, the heartwood compartment increases when sapwood dies, but its own mortality rate is zero. The updating process is a simple mass balance:

$$B_i(t) = B_i(t-1) + G_i(t-1) - M_i(t-1) \quad (7.27)$$

i denotes the different compartments, B , G and M are the biomass, growths and mortalities, respectively. The leaf area index (LAI) is necessary for calculation of photosynthesis and a number of micro-climatic estimations. It is connected to foliage biomass by the specific leaf area for each canopy layer (LAI^{spe} (eq. 7.49))

$$LAI = B_{foliage} \cdot LAI^{spe} \quad (7.28)$$

The development and senescence of foliage depends on the temperature sum above a certain minimum value as in the PnET-model (Aber *et al.* 1996). All common plant species demonstrate a linear relationship between foliar nitrogen concentrations and CO_2 -assimilation rate which makes the photosynthesis-N relationships a very useful approach in modelling (Aber *et al.* 1996).

$$A_{max} = A_{max_A} + A_{max_B} \cdot N_{foliage} \quad (7.29)$$

Since the enzymatic speed of the photosynthesis process, A_{max} depends on the nitrogen concentration in the foliage the lower assimilation rate for old needles is accounted for as they have lower nitrogen content. A_{max_A} and A_{max_B} are parameters specific for either deciduous or coniferous trees (Reich, Kloeppel and Ellsworth 1995). The maximum photosynthesis rate is considered limited by three factors, temperature, vapour pressure deficit and water stress (see eq. 5.14). The temperature dependency of the assimilation rate, D^{temp} , is derived from an

optimum function suggested by Bossel (1994).

$$D^{temp} = \begin{cases} \left(2 \cdot \frac{T_{air_day}^{avg} - T_A^{min}}{T_A^{opt} - T_A^{min}} \right)^2 & \text{if } T_{air_day}^{avg} < T_A^{opt} \\ 1 - SLO_{AT} \cdot \left(\frac{T_{air_day}^{avg} - T_A^{min}}{T_A^{opt} - T_A^{min}} \right)^2 & \text{if } T_{air_day}^{avg} \geq T_A^{opt} \\ 0 & \text{if } T_{air_day}^{avg} < T_A^{min} \end{cases} \quad (7.30)$$

The effect of vapour pressure deficit (air dryness), VPD , is calculated with a power function (Aber *et al.* 1996).

$$D^{VPD} = 1 - D_1^{VPD} \cdot VPD D_2^{VPD} \quad (7.31)$$

The maximum photosynthesis rate are not maintained throughout the entire day. To get a daily average value the factor A_{max}^{frac} is set to 0.76 (Aber *et al.* 1996). Hence, the gross assimilation (including basal respiration of leaves, see paragraph ??) is calculated

$$A_{max}^{gross} = \left(A_{max} \cdot A_{max}^{frac} + Resp_{foliage} \right) \cdot D^{VPD} \cdot D^{water} \cdot D^{temp} \quad (7.32)$$

Basal leaf respiration is calculated as a fraction of A_{max} .

$$Resp_{foliage} = A_{max} \cdot Resp_{foliage}^{frac} \quad (7.33)$$

Daytime and nighttime leaf respiration is calculated separately with a Q_{10} -function.

$$\begin{aligned} Resp_{foliage}^{day} &= Resp_{foliage} \cdot RespQ_{10}^{((T_{day}^{avg} - T_A^{opt})/10)} \cdot DAYL \\ Resp_{foliage}^{night} &= Resp_{foliage} \cdot RespQ_{10}^{((T_{night}^{avg} - T_A^{opt})/10)} \cdot NIGHTL \end{aligned} \quad (7.34)$$

The same temperature dependency is used for calculation of maintenance respiration of the other living compartments of the tree. This respiration is also dependent on the nitrogen and mineral contents of the compartment as suggested by Penning de Vries (1989). It is also reduced if the freely available carbohydrates are below a certain threshold. The dependence on physiological activity is accounted for by adding a certain percentage of photosynthetic production distributed to each compartment (Grote and Suckow 1998). Growth respiration is calculated as a constant fraction of available carbohydrates for allocation, see section 7.2.5.

The mortalities in the living compartments are calculated by multiplying a biomass base value with a mortality function that depends on environmental conditions. This procedure is used for all compartments except regenerative tissue that dies completely within a given time frame.

$$M_i = B_{M_i}^{ref} \cdot F_{M_i} \quad (7.35)$$

$B_{M_{foliage}}^{ref}$ is the biomass at the beginning of the year. The mortality factor for foliage is based on a longevity parameter specific for each age class and a mortality development state calculated from the sum of all monthly respiration and assimilation. $B_{M_{fineroots}}^{ref}$ is the actual mass of the compartment. The mortality rate depends on the maximum and minimum daily turnover rates and a water stress factor. The water stress factor is derived from available water, field capacity and wilting point. The daily turnover rates for fine roots are poorly known, and should be further investigated.

$$F_{M_{fineroots}} = M_{fineroots}^{max} - (M_{fineroots}^{max} - M_{fineroots}^{min}) \cdot \left(1 - \left(\frac{W_{soil}^{fc} - W_{soil}}{W_{soil}^{fc} - W_{soil}^{wp}} \right)^2 \right) \quad (7.36)$$

The sapwood mortality is derived from the difference between last years actual sapwood mass and sapwood demand. The actual sapwood mass depends on the amount of carbon allocated

the previous year and the sapwood demand depends on the annual height growth and the upwards shift of the crown. No explicit function for sapwood mortality is implemented but the distribution is supposed to equal that of foliage mortality. The mortality for coarse roots and branches is derived the same way. The mortality of regenerative tissue is calculated using the same concept as for foliage, but the process is supposed to be 1.5 times faster.

7.2.5 Allocation and growth

Research at the process level in carbon allocation (and fine root dynamics, for that matter) has been sadly neglected to the benefit of research on photosynthesis, respiration and leaf growth. The lack of understanding of the carbon allocation mechanisms is now the limiting factor in process-based modelling of whole trees (Cannell and Dewar 1994). One of the core issues in the SUFOR project is sustainable forestry under changing environmental conditions. These conditions include climate change and changes in anthropogenic impact such as pollution and management. All of the above have substantial impact on both carbon and nitrogen allocation (Grote 1998b). Thus, the carbon allocation model should consider:

- Climatic factors
 - Temperature
 - Water supply
 - Light
 - CO₂ supply
- Nutrient supply
 - Nitrogen
 - Phosphorous
 - Potassium
 - Magnesium
 - Manganese

The work is started to develop a new allocation model that will consider as much as possible of the above. The allocation algorithm used in the present version of ForSAFE is based on a sink strength concept (Grote 1998b). The sink strength of a certain plant compartment, D_i^{rel} , is the amount of carbon the compartment lacks to be of optimum size.

$$D_i^{rel} = \max\left(0, \frac{B_i^{opt} - B_i}{B_i^{opt}}\right) \quad (7.37)$$

The carbon available for growth equals assimilated carbon minus total respiration. The growth respiration is almost the same in all kinds of tissue and the loss can be described by a constant factor.

$$Pool = \left(A^{tot} - \sum RES_i - RES_A\right) \cdot F_G \quad (7.38)$$

The growth of each compartment is calculated according to their specific sink strength.

$$G_i = POOL \cdot \frac{D_i^{rel}}{\sum_i D_i^{rel}} \quad (7.39)$$

The optimum sizes of the compartments are derived from their functional balance ratio (see eq. 7.40) to foliage biomass. These relations will only be briefly discussed here, the interested reader is directed to Grote (1998, 1998b).

$$FR_{sap} = \frac{B_{foliage}}{(B_{cone} - B_{core}) \cdot (1 + F_{coarse} + F_{branch})} \quad (7.40)$$

B_{cone} and B_{core} are the biomass of sapwood and heartwood, respectively (see section 7.2.5). F_{coarse} and F_{branch} are species specific parameters that express the coarse root and branch biomass relation to sapwood. Reserve biomass (freely available carbohydrates) is part of other living tissue and therefore impossible to measure and thus validate. Two assumptions are made:

- All reserves are stored in the foliage and the sapwood.
- The carbon store capacity is equal to the optimal biomass relation.

The optimum reserve biomass relates to foliage and sapwood biomass by the sum of two demands; the first is the need to fill the storage capacity with carbohydrates needed for flushing and the second is activated during growth of regenerative tissue and represents carbohydrates needed to build new regenerative tissue. The allocation to foliage is determined by the phenological status, which develops cumulatively, and a predetermined amount of foliage biomass, calculated from the actual foliage mass at the beginning of the year, the maximum foliage mass and the fraction of expected foliage loss until the end of flushing. The allocation to fine roots is determined by the ratio of optimum fine root mass to foliage mass. This ratio increases with increasing demand for water and nutrients (only nitrogen, at this stage). The allocation to sapwood depends on tree dimensions. The required ratio between sapwood and foliage also determines demand for branches and coarse roots because of fixed (coarse roots) or stem diameter dependent (branches) relations. Nitrogen uptake follows the process formulations discussed in the process chapter.

Carbon and nitrogen budgets

The decomposition submodel DECOMP accounts for carbon and nitrogen budgets in the soil in ForSAFE (Walse 1998). The model operates with reservoirs, see Fig. 7.14, of organic matter and rate equations for their change. These have been formulated as kinetic equations, depending on state variables and modified by boundary conditions. The decomposition model DECOMP is discussed in detail in the process chapter. In polluted areas the uptake through the canopy is a significant source of nitrogen (Lovett and Lindberg 1984, Skeffington and Wilson 1988). The higher deposition in forested areas compared to open fields (Bergmann and Flöhr 1989) is accounted for and depends on leaf area and stand density. If historic deposition data is missing the deposition history can be reconstructed with the MAKEDEP model (Alveteg, Walse and Warfvinge 1998). It is a tool developed for reconstructing historic atmospheric deposition for forests using present days values. The basic principle of the reconstruction model is to scale deposition and emission trends in the literature to fit measurements of atmospheric deposition at the studied site. The model divides the the deposition of compounds into into wet and dry categories and scales them with canopy size. The dividing into wet and dry deposition is necessary because it is mainly dry deposition that is affected by vegetation change. The marine and non-marine fractions are necessary to handle separately at sites close to the sea since sea spray give these sites a characteristic deposition pattern. Standard curves for NO_3^- , NH_4^+ and SO_4^{2-} deposition for Northern Europe have been derived for the period 1800-2000 (Alveteg *et al.* 1998). These curves are used to scale the non-marine deposition, in case of dry deposition together with the canopy biomass relationship. see Table 7.2

| Deposition | Element | Proportional to canopy biomass | Proportional to standard curve |
|-----------------|---|--------------------------------|--------------------------------|
| Dry marine | BC, Na ⁺ , Cl ⁻ , SO ₄ ²⁻ | yes | no |
| Wet marine | BC, Na ⁺ , Cl ⁻ , SO ₄ ²⁻ | no | no |
| Dry non-marine | BC, Cl ⁻ , SO ₄ ²⁻ | yes | SO ₄ ²⁻ |
| | NO ₃ ⁻ | yes | NO ₃ ⁻ |
| | NH ₄ ⁺ | yes | NH ₄ ⁺ |
| Wet non-marine | BC, Cl ⁻ , SO ₄ ²⁻ | no | SO ₄ ²⁻ |
| | NO ₃ ⁻ | no | NO ₃ ⁻ |
| | NH ₄ ⁺ | no | NH ₄ ⁺ |
| Canopy exchange | BC | yes | no |

Table 7.2: Reconstruction of different deposition categories. BC are base cations Ca²⁺, Mg²⁺ and K⁺.

PnET yearly processes, building of the stand

The stand properties are calculated once a year. The annual changes in the stand are calculated separately for each social tree class. First of all the increase in height and diameter is calculated, as mentioned above. Then the other dimensions and stand properties are updated. The functional balance ratio (FR_{sap} , see eq. 7.40) between foliage and sapwood, mentioned in the allocation chapter, cannot be considered constant because of the increasing distance between roots and crown. From sapwood growth of the previous year, the height and diameter increase is derived and other stand properties have to be updated considering that. Average stand height, H_{avg} , stem volume without bark, V_{stem} and diameter at breast height, DBH are obligate initialisation variables.

The number of trees per hectare is calculated from these initialisation variables and the reverse equation yields the stem volume every year, considering new tree dimensions and the decreased number of stems.

$$n = \frac{V_{stem}}{H_{avg} \cdot D_0^2 \cdot \frac{\pi}{4} \cdot \frac{1}{3}} \quad (7.41)$$

The total annual sapwood growth of the stand is divided by the number of trees in the stand to get the average stem growth. Diameter and height growth are calculated as suggested by Bossel (1994).

$$\Delta D_0 = \frac{4 \cdot G_{stem}}{DENS \cdot \pi \cdot Q_{HD}^{opt} \cdot D_0^2} \quad (7.42)$$

Q_{HD}^{opt} is an optimum height/diameter ratio related to crown area index, $CRAI$. The normal stand density (number of stems per area unit) does not take into account the width of the branches and is thus insufficient to estimate changes in allocation and growth after changes like thinnings or other disturbances in the stand. Also the impact of canopy on throughfall and light penetration are better estimated if the crown area index is used. $CRAI$ describes the vertical projection area of the tree crowns per hectare (Bossel 1994). The ratio between crown diameter and DBH , Q_{CD} is used as well as maximum and minimum height/diameter ratios (species specific parameters).

$$CRAI = \left(DBH \cdot Q_{CD} \cdot \frac{(Q_{HD}^{max} + Q_{HD}^{min})}{2 \cdot Q_{HD}} \right)^2 \cdot \frac{\pi}{4} \cdot \frac{n}{10000} \quad (7.43)$$

When the diameter growth is calculated, the height growth, ΔH^{avg} , can be determined.

$$\Delta H^{avg} = \Delta H^{avg^{max}} \cdot \left(1 - \frac{\Delta D_0}{\Delta D_0^{max}} \right) \quad (7.44)$$

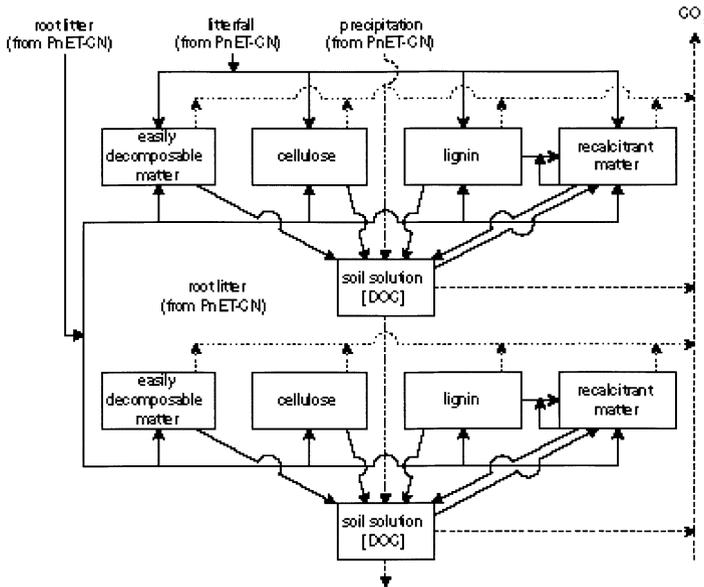


Figure 7.14: Flowchart of the two uppermost layers in the DECOMP model. The second layer is repeated in layers three and four.

$$\Delta D_0^{max} = \frac{6 \cdot G_{stem}}{DENS \cdot \pi \cdot D_0 \cdot H_{avg}} \quad (7.45)$$

$$\Delta H^{avg^{max}} = \frac{12 \cdot G_{stem}}{\pi \cdot D_0^2} \quad (7.46)$$

The trees in the model are represented as a cone for stem and a cylinder for crown, see Fig. 7.16. The stem is divided into sapwood and heartwood, where the heartwood constitutes of a core cone inside the stem cone.

The foliage biomass is obtained by dividing LAI by average specific leaf area.

$$B_{foliage} = \frac{LAI}{LA_{spe^{avg}}} \quad (7.47)$$

The potential LAI is given in the initialisation process and the median of the specific leaf area is given as a parameter. The actual LAI is calculated from potential LAI reduced by the expected growth of the first age class and $CRAI$, until maximum foliage is reached. About a fourth of the needles in the first of the age classes is supposed to die during the expansion phase and hence, the starting LAI must be increased.

$$LAI = LAI^{pot} \cdot CRAI \cdot \left(1 + 0.25 \cdot \frac{F_{age(1)}}{1 + \sum F_{age(a)}} \right) \quad (7.48)$$

The specific leaf area is calculated from specific leaf areas of each canopy layer, derived from empirical investigations (Kellomäki and Oker-Blom 1981). The developed function needs the median of specific leaf area and its dependency of canopy depth as parameters and is therefore easily parameterized for different species or age classes. The average specific leaf area is obtained by weighting the specific leaf area in each layer with the relative foliage distribution (eq. 7.50).

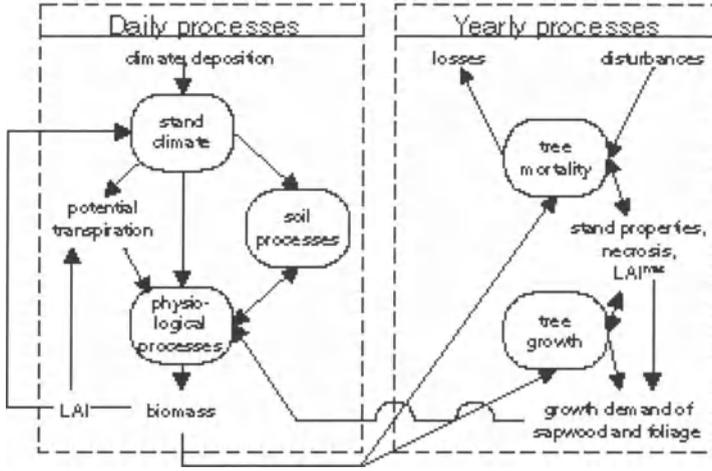


Figure 7.15: Interactions between daily and yearly processes.

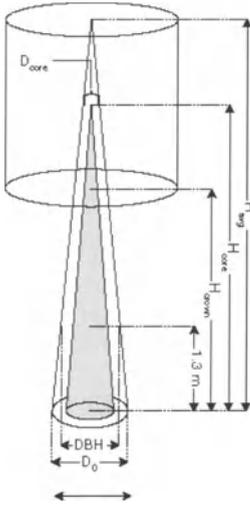


Figure 7.16: The stem in the model is represented by two coaxial cones, the sapwood cone and the heartwood cone. The crown is represented by a cylinder.

of sapwood while the branch wood fraction ($F_{branches}$) is developing according to a relation with stem diameter and crown diameter (Bossel 1994). The sapwood and heartwood fractions of branches and coarse roots are considered equal to the relationship in the stem.

$$F_{branches} = F_{branches}^{min} + (1 - F_{branches}^{min}) \cdot e^{(-10 \cdot \frac{DBH}{DBH^{max}})} \quad (7.54)$$

$$LA_h^{spe} = LA^{spe_{med}} \cdot 2^{\frac{(\frac{h}{nh} - 0.5) \cdot LA^{spe_{med}}}{SLO_{LA}}} \quad (7.49)$$

$$LA^{spe_{avg}} = \sum_1^{nh} (LA_h^{spe} \cdot F_{foliage_h}) \quad (7.50)$$

There are four compartments of woody biomass, sapwood, heartwood, branchwood and coarse roots. The fractions of heartwood and sapwood in the stem are determined as the core of the cone and the outer shell of the cone, respectively.

$$B_{cone} = \frac{H^{avg} \cdot D_0^2 \cdot \pi \cdot DENS}{12} \quad (7.51)$$

$$B_{core} = \frac{H_{core} \cdot (D_0 - D_{core})^2 \cdot DENS \cdot \pi}{12} \quad (7.52)$$

$$H_{core} = H_{crown} + F_{crown} \cdot \frac{LAI^{max}}{LAI^{pot} \cdot CRAI} \cdot (H^{avg} - H_{crown}) \quad (7.53)$$

The peak of the core inside the cone is calculated to take into account the changes in foliage biomass. That is because sapwood demand is dependent on the foliage biomass that has to be supported. This is implemented by weighting the variable for relative core height within the crown (F_{crown}) with the degree to which the potential LAI was actually reached the previous year. The coarse roots are considered a constant fraction of sapwood while the branch wood fraction ($F_{branches}$) is developing according to a relation with stem diameter and crown diameter (Bossel 1994). The sapwood and heartwood fractions of branches and coarse roots are considered equal to the relationship in the stem.

The fine root distribution is an object of current research within SUFOR. The results from this investigation and the new allocation algorithm that is being developed will be the basis of a new algorithm for fine root distribution.

The annual reduction of the number of stems is due to either a vitality function or thinning events. The vitality index characterises tree health and is calculated by dividing annual net production by total annual mortality of the compartments, including exudation losses.

$$VI = \frac{\sum POOL}{\sum (M_i)} \quad (7.55)$$

The removed fraction of biomass is calculated with a slope parameter, SLO_{VI} .

$$F_{VI} = \begin{cases} 1 - VI^{SLO_{VI}} & , \text{ if } VI < 1 \\ 0 & , \text{ if } VI \geq 1 \end{cases} \quad (7.56)$$

All biomass compartments, nitrogen pools and stem number are reduced by equal proportions, meaning there will be no change in average tree dimensions. If thinnings occur, a disturbance routine replaces the mortality function. The thinning does not randomly select trees. Whenever a defined height has been reached or the height growth interval exceeds a certain level, stems will be removed. The intensity of the thinning is described by PSN , a parameter that can be set differently for each thinning event. The stems removed can be set as bigger or smaller than the average stem by varying the parameter $THIN$.

$$F_n = 1 - e^{(-PSN \cdot THIN \cdot \Delta H^{avg})} \quad (7.57)$$

$$F_{VI} = 1 - e^{(-PSN \cdot \Delta H^{avg})} \quad (7.58)$$

7.3 Decomposition of organic matter

7.3.1 Introduction

This model is a part of the efforts needed to build the model tool being developed for the Sustainable Forestry in Southern Sweden Program (SUFOR), the ForSAFE model. This important model involve a tree growth and physiology submodel of the PnET-FORSANA type (Aber and Melillo 1995, Fichter 1999), a geochemical submodel derived from SAFE/PROFILE and a carbon-nitrogen submodel which is described here. This submodel will be included in the ForSAFE at a later stage. The scope was to computerize the Walse-Berg-Sverdrup model (Walse et al., 1998) and to explore the properties of such an integrated model. This rests on a long term goal of enhancing the SAFE model with components for carbon cycling and mechanistic growth of trees to develop ForSAFE. We want to explore the results with respect to the standard reference behaviour and explore the dynamics of this model. This is the preparation necessary before this model can be integrated into our larger forest ecosystem model under development.

The mechanism concept presented in this study is based on the work of Mindermann (1968) and Tenney and Waksman (1929). Our model concept has basic characteristics in common with several others, such as the models described by Bunell et al. (1977b), McGill et al. (1981), Parton et al., (1987), Molina et al. (1983), Parnas (1976) and van Veen et al, (1984), as well as developments by the authors (Walse et al. 1998). These models are all process-oriented in that they operate with the separate influence of environmental factors such as temperature and moisture on the decomposition process. They differ in time resolution of calculations, the type of ecosystem modeled and in the number of different substances accounted for (carbon, nitrogen, phosphorus, etc.). The models differ also in the number of pools of organic matter accounted for. Moisture and temperature are frequently accounted for in decomposition models. The idea that pH would affect decomposition rate is well established and certain models have included an impact of pH on decomposition rate, e.g. the NICCCE model van Dam and van

Breemen (1995) but to our knowledge this is the first time such a large selection of experimental data is synthesized into rate-regulating model functions. A number of factors may influence decomposition, and they may be classified into categories such as:

1. External conditions
 - (a) Water
 - (b) Temperature
 - (c) Soil solution chemistry
 - i. Soil acidity
 - ii. Concentration of calcium, magnesium and potassium
 - iii. Concentration of heavy metals and toxic contaminants
 - (d) Inorganic oxidation
2. Substrate quality
 - (a) Substrate type
 - (b) Nitrogen concentration in organic matter
 - (c) Phosphor concentration in organic matter
 - (d) Manganese and iron concentration in organic matter
 - (e) Heavy metal concentration in organic matter
3. Microbial conditions
 - (a) Composition and relative importance of fungal community
 - (b) Composition and relative importance of bacterial community
 - (c) Composition and relative importance of heterotrophic community

7.3.2 Objectives

The objective is to review and synthesize some of the qualitative and quantitative information available on the relationships between organic matter decomposition rate and soil chemistry and environmental factors. The principal objective is to outline and parameterize a decomposition submodel. The purpose of this study was to compile a unified data set for a range of soils that have been used for decomposition experiments by using previously published data and convert them. This was achieved by estimating some unreported background data from other models or methods of estimation. The decomposition data were then evaluated using a variety of moisture indices, including soil matric potential, soil moisture content relative to field capacity and volumetric soil moisture content.

Our aim was to find a model function valid for all soils suitable for use in a more complex decomposition model, that would also include the effect of temperature and possibly other environmental parameters such as soil acidity or heavy metal concentration. The scope was twofold: (1) find which moisture index that most accurately describes the relation between soil moisture and decomposition; and (2) compare the moisture indices from a data requirement point of view, to find out whether there is a moisture index that with a minimum of measurements reproduces the relation decomposition-moisture with an acceptable accuracy. A model based on such a moisture index would be very valuable in the context of regional/global decomposition modelling, where data availability is limited. The range of soil moisture conditions which we considered were dry to nearly saturated soils, as decomposition of organic material in saturated soils is governed by different processes from the ones in unsaturated soil.

The scope of this is to review the effects of temperature, moisture, soil solution acidity and soil solution concentration of base cations (Ca^{2+} , Mg^{2+} and K^{+}) on decomposition rate. The model is intended to be applied primarily to forest soils in Northern Europe. It is recognized that other factors than the here studied may be important for the decomposition rate. Some of the more studied such factors are substrate quality, heavy metal concentration and concentration of phosphorus and nitrogen in soil solution. An important assumption in this conceptual model is that the soil microorganisms are the fully responsible for decomposition. Larger soil animals are of some importance for the decomposition of organic material,

but bacteria and fungi have been estimated to cause 80% of the carbon circulation (Wood 1989). For nutrient-poor northern European forests, Persson et al. (1980) estimated the corresponding figure to 90%. Therefore, the influence of the larger soil animals is considered to be mainly fractionation. Moreover, earth worms, one of the more important groups of soil animals in this sense, occur sparsely in the acid podsols that are of most interest for this study. The biomass of bacteria and fungi is in this paper referred to as the *microbial biomass*. An important simplification in the model is that soil solution chemistry and climatic factors are affecting the activity and functional composition of the microbial biomass. The soil is regarded as entirely homogeneous, so that the difference between conditions at micro-sites when compared to conditions in the bulk solution are ignored. The division of the soil into several layers is an important feature of the geochemical model (SAFE) for which the decomposition submodel is developed. At present, however, the decomposition model is developed for a single soil layer, corresponding to the litter layer plus the organic horizon.

7.3.3 Model concept

The model is in its present state formulated for the transformation of carbon only. Figure 7.17 shows a conceptual view of the model. In nature, the laws of competition will allow an increase in one species due to a superior property in a given situation, when another species competing for the same ecological niche is weakened or eliminated. This reasoning is applicable also for the microbial biomass. In most cases many different species, possibly with different environmental requirements, can cause the same or similar chemical transformations. A few soil processes are quite specific, however, and may only be performed by one or a few species. For example, nitrification is believed to be performed by two specific bacteria, nitrosomonas and nitrobacter (Wood 1989). The decomposition of the lignin and the resistant material, is believed to be more specialized than the decomposition of the easily decomposable material and of the holocellulose. The white-rot fungus *Coriolus versicolor* and the soft-rot fungi *Chaetomium globosum* are two well investigated species of lignin decomposers. Today we may see at least 40 species of fungi that may degrade lignin completely (Eriksson et al. (1990)). It is possible that bacteria also play

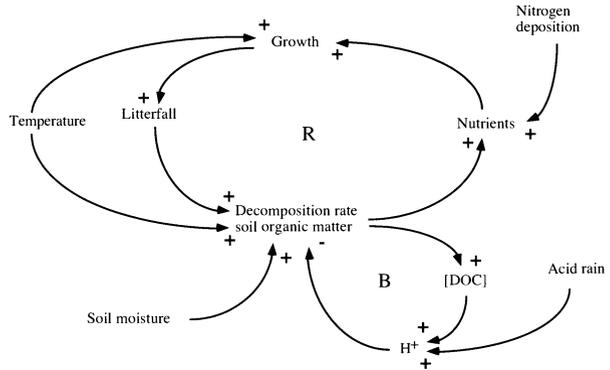


Figure 7.17: Causal loop diagram of the decomposition process in the soil. This represents our mental understanding embedded in the Walse-Berg-Sverdrup concept for the decomposition dynamics in the soil. The system is characterized by several feedback loops. The internal reinforcing loops have additional balancing loops, preventing the system from escalation. The causal loop diagram represented here is applied in the computer model used later in this study.

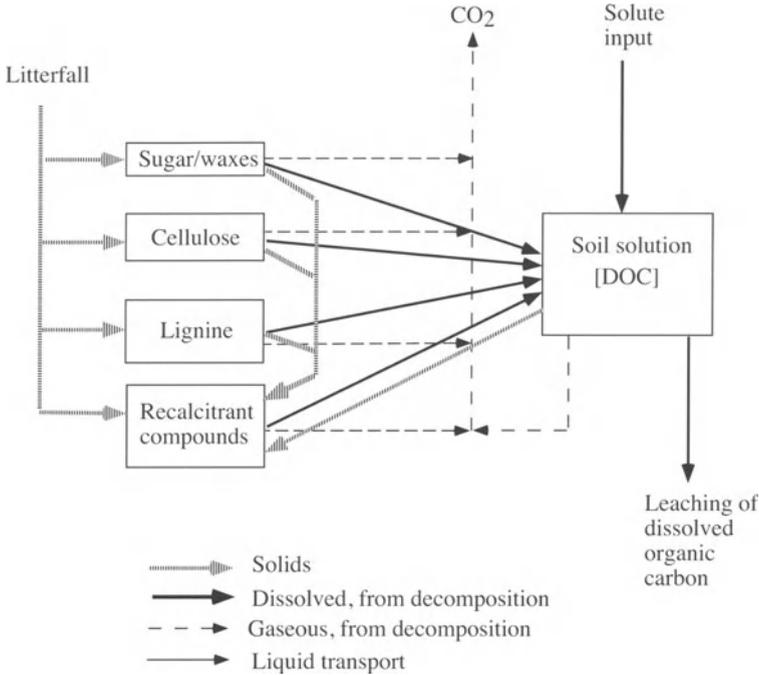


Figure 7.18: Flow pattern of carbon in the soil according to the Walse-Berg-Sverdrup model. Important is the division of new organic matter into four types and the feedback from soil chemistry, temperature and soil wetness. This scheme was applied in each soil layer.

a part in the degradation of lignin, but it is not known to what degree they contribute to the lignin decomposition rate (Kirk and Zeikus 1978). We hypothesize that total decomposition rate will be limited by the activity of bacteria and fungi. It is also assumed that fungi are relatively more important for the decomposition of lignin and resistant compounds while bacteria are more important for the decomposition of the easily decomposable material. The decomposition of the substances in the holocellulose pool is assumed to depend equally on bacterial and fungal activity. The relative importance of fungi and bacteria for the decomposition of a certain soil organic matter pool can then be defined as:

$$\text{Rel. BD} = \frac{r_{BD}}{r_{BD} + r_{FD}} \quad (7.59)$$

and

$$\text{Rel. FD} = \frac{r_{FD}}{r_{BD} + r_{FD}} \quad (7.60)$$

where r_{BD} denotes bacterial decomposition and r_{FD} denotes fungal decomposition ($\text{g m}^{-2} \text{ yr}^{-1}$). Specific rate regulating factors for fungal and bacterial growth can be separated in some cases, as will be shown further on. As a first assumption, we will go even further and assume fungi to be solely responsible for the decomposition of lignin and resistant material, whereas bacteria is assumed to be solely responsible for the decomposition of the easily decomposable material (table 7.3.5). Since the lignin and the resistant material pools both are assumed to be decomposed by fungi only, the rate regulating functions will be the same for these two pools. The only difference in model parameters between these two pools is thus the potential rate constants

(k_{pot_3} and k_{pot_4}). For each organic matter fraction i we have the kinetic rate expression:

$$r_i = k_i \cdot m_i \cdot f(w) \cdot g_i(pH) \cdot j_i(T) \cdot z(N) \cdot v(M) \quad (7.61)$$

The action of soil chemistry is expressed either by a simplified expression;

$$g(pH) = \frac{1}{1 + k_H \cdot [H^+]^n} \quad (7.62)$$

or a more elaborate, the "full expression";

$$g(BC/Al/H) = \frac{[BC^{2+}]^m}{[BC^{2+}]^m + \kappa_M \cdot ([Al^{3+}] + 3 \cdot [H^+])^n} \quad (7.63)$$

The action of water is expressed using;

$$f(w) = \frac{K_W \cdot w^v}{1 + K_W \cdot w^v} \quad (7.64)$$

and the effect of temperature is expressed by using an Arrhenius expression:

$$j(T) = r_0 \cdot 10^{A/(273+T) - A/281} \quad (7.65)$$

7.3.4 Data and Methods

Data was obtained from the literature and analyzed on the basis of the assumed rate-regulating functions. The maximum relative reaction rate was set to 1.0 at optimal conditions, which were assumed to be: (i) moisture content at field capacity, (ii) pH 6 and (iii) maximum BC^{2+} to $(Al^{3+} + p \cdot H^+)$ ratio in each experiment, provided that it was above 1.0. Experimental data include a variety of measurement techniques and materials used, from mass loss rates in field litterbag experiments measured for up to several years to laboratory experiments of bacterial growth rates. The common factor of the data collected is the ability to tell us something about how decomposition rates may be affected by the three selected environmental parameters temperature, moisture or acidity. None of the experiments include all three of these parameters, however, some experiments include two of them, and some include only one. The activation energy was obtained as the slope when plotting $\ln(k)$ versus $-1/T$. References and key information on the experiments are listed in table 7.3. The model parameters for the rate regulating functions other than the one for temperature (i.e. functions g_1 - g_4 and θ_1 - θ_4) were obtained through the use of a computerized curve fit procedure. References and key information on the experiments are listed in tables 7.7, 7.12 and 7.11. Experimental data on the relation between pH and Al on microbial process rates were divided in two; (i) experiments including measurements of pH, Al and base cation concentrations and (ii) experiments where only pH was measured. The former group were used to extract model parameters for the "full" ion exchange expression (7.63) and the latter group was used to extract model parameters for the simplified ion exchange expression. The ion-exchange constant K and the n and m parameters in the "full" ion exchange function were obtained by analyzing data on bacterial growth and fungal growth as dependent on pH, Al and BC concentration in several steps. First, an empirical function of the same type as equation 7.63 was fitted to the data, letting the n and m exponents vary freely to obtain the best possible fit. This was done using the data from each experiment, to investigate whether the n and m exponents were in the range of the ones assumed in the Vanselow, the Gapon and the valence unspecific ion exchange functions;

| Mechanism | n | m | p |
|--------------------|-----|------|---|
| Valence unspecific | 1 | 1 | 1 |
| Gapon | 0.5 | 0.33 | 1 |
| Vanselow | 3 | 2 | 3 |

It also gave the maximum correlation to be expected when applying these theoretical ion exchange expressions. It was repeated for p values ranging from 0 to 6, to find out whether the theoretical p values assumed were appropriate. Thereafter, the three theoretical ion exchange functions were applied to data from each experiment and the correlations between data and function were evaluated. Finally, the experimental data were grouped according to the measured process (bacterial growth or fungal growth) and model parameters were calculated using all data within each group in combination with the selected ion exchange function. For the simplified ion exchange expression, only the valence unspecific ion exchange function was evaluated (simplified version of Eq. 7.63). An empirical function of the same type as equation 7.62 was fitted to the data, letting m vary freely to obtain the best possible fit. This empirical function was then compared to the valence unspecific ion exchange function. A serious problem in this type of analysis is that all experiments were not performed over the whole range of environmental conditions. If, for example, a moisture experiment is performed over a range of 0-50% of field capacity, the highest relative process rate should not be scaled to 1, but to somewhere between 0 and 1. This kind of data is almost impossible to use, as long as one does not have another experiment, involving the same process and using the same method, to use as a guide. In the case of $BC^{2+}/(Al^{3+} + p \cdot H^+)$ values, this problem could not be ignored, as no experiment was performed over exactly the same experimental interval. Therefore, the maximum $BC^{2+}/(Al^{3+} + p \cdot H^+)$ value in each experiment was assumed to allow maximum growth, provided that it exceeded 1.0. Finally, potential rate constants (rate constants at optimal conditions) were calculated from a number of decomposition experiments. This was done by applying the rate regulating functions for temperature, moisture and pH found in this study to a number of field experiments where pH, soil moisture and temperature were all available.

7.3.5 Results

Temperature

Data on the impact of temperature on decomposition rates is abundant and only a small selection of the available data is reported here. Three different experimental methods are frequently found in the literature: measurement of the mass loss of organic material, measurement of nitrogen mineralization (measured as increase in ammonium and nitrate concentration) and respiration measurements. The reported experiments can be divided into two groups, medium and long-term experiments with measurements lasting from 2 weeks up to 2 years, and short-term experiments with measurements for up to 24 hours. The short-term experiments are all field experiments using different types of respirometers, while the main part of the medium and long-term experiments are laboratory experiments of different types. The short-term experiments show a much stronger response to temperature fluctuations. The longer experiments, irrespective of method, give a lower activation energy (table 7.3). The short-term respiration measurements are here assumed to mirror immediate responses in microbial activity, while the more long-term experiments are assumed to show the integrated effect of temperature on the soil processes leading to decomposition. Another reason for the difference between the two groups may be that the short-term experiments are performed in the field, where carbon dioxide from root respiration is added to that from the decomposition. Model parameters were based on the results from the medium- and long-term experiments. The range in the data from medium- and long-term experiments reflect differences in material composition and duration of the experiment. It is difficult to resolve the results further, but a slight trend in data is seen with respect to the substrate used. The data from Jenny et al.(1949), suggests that the decomposition of more easily-decomposed material, in this case alfalfa, is less temperature-dependent than the decomposition of cellulose (Donnelly et al. 1990; Rosswall 1974; Rosswall and Berg 1972). Moreover, decomposition rates of pure lignin or humus Donnelly et al 1990; Mork 1938; Sohlenius et al 1976) show a stronger temperature dependence. The model parameters were calculated as the average from those experiments where material corresponding as closely as possible to the

| Reference | Material | Method | Duration | E_a (kJ mol ⁻¹) |
|----------------------------------|-------------------|-----------------------|------------|----------------------------------|
| Med. & long-term exp. | | | | |
| Easily decomp.: | | | | |
| Jenny et al 1949 | Alfalfa | Mass loss | 1 yr | 17.9 |
| Holocellulose: | | | | |
| Rosswall 1974 | Cellulose | Mass loss | 0.3-1 yr | 38.8 |
| Donnelly et al 1990 | Cellulose | CO ₂ prod. | 6 weeks | 59.6 |
| Donnelly et al 1990 | Cellulose | Mass loss | 8 weeks | 56.9 |
| Lignin: | | | | |
| Donnelly et al 1990 | Lignin | CO ₂ prod. | 6 weeks | 54.8 |
| Humus: | | | | |
| Mork 1938 | Humus | N min. | 8-50 weeks | 52.3 |
| Mork 1938 | Humus | CO ₂ prod. | 4-16 weeks | 48.4 |
| Sohlenius et al 1976 | Humus | N min. | 14 weeks | 60.2 |
| Mixed pools: | | | | |
| Wood 1974 | Eucalyptus litter | Mass loss | 1 yr | 30.6 |
| Theodorou and Bowen 1983 | Pine needles | N min. | 8 weeks | 38.2 |
| Berg et al 1991 | Pine needles | Mass loss | 1-2 yrs | 57.2 |
| Sowden and Ivarson 1974 | Mixed litter | Mass loss | 4 yrs | 31.7 |
| Sowden and Ivarson 1974 | Mixed litter | Mass loss | 2.5 yrs | 34.9 |
| Sowden and Ivarson 1974 | Mixed litter | Mass loss | 2 months | 56.3 |
| Stanford et al 1973 | Mixed litter | N min. | 2 weeks | 45.3 |
| Chen et al 1973 | Mixed litter | Mass loss | 1-2 years | 54.4 |
| Cassman and Munns 1980 | Soil | N min. | 2 weeks | 31.8 |
| Campbell and Biederbeck 1975 | Mineral soil | N min | 2 weeks | 63.9 |
| Macduff and White 1985 | Mineral soil | N min. | 10 days | 37.1 |
| Short-term exp. | | | | |
| Flanagan and Veum 1974 | Tundra | CO ₂ prod. | 1 hr | 93.7 |
| Anderson 1973 | Beech leaves | CO ₂ prod. | 1-2 hrs | 64.6 |
| Anderson 1973 | Chestnut leaves | CO ₂ prod. | 1-2 hrs | 69.9 |
| Edwards 1975 | Forest floor | CO ₂ prod. | 24 hrs | 110.9 |
| Buyanovski and Wagner 1983 | Forest floor | CO ₂ prod. | Instant | 83.8 |
| Castelle and Galloway 1990 | Forest floor | CO ₂ prod. | Instant | 84.9 |

Table 7.3: *Relationship between temperature and decomposition rate, expressed as the activation energy. Model parameters were calculated using all data representative for each pool. Data from mixed pools can be used for parameterization of a single-pool model. Data from short-term experiments are shown as a comparison.*

defined model pools were used (tables 7.3). For a simpler model, using only one pool of organic material, the appropriate value of E_a would be 45 kJ mol⁻¹, based on data from decomposition of mixed litter (table 7.3).

Moisture

Decomposition of organic matter is of great interest, especially in the light of a possible global warming and the subsequent increased carbon dioxide (CO₂) fluxes that may follow, but also in the context of nutrient cycling in ecosystems. The soil decomposers need water for their metabolism, and thus this affects their activity level. It is hypothesized that the effect of soil moisture depend on the wetting of the soil material and wetting of the surfaces of the soil decomposer organisms. It would then be reasonable to assume that this would follow a Freundlich adsorption isotherm for physical adsorption of water to a particulate solid (Sverdrup et al, 1992). Full moisture saturation could lead to anoxic conditions, and decomposition would stop completely (Burl, 1985; Swift et al., 1987). Numerous studies have investigated this relationship for specific soils (Andren et al 1992, Campbell and Biederbeck 1975; Quemada and Cabrera 1997) and for a range of different soils (Stanford and Epstein 1974; Miller and Johnson 1964; Myers et al 1982). The basis of such studies are usually CO₂ evolution, nitrogen (N) mineralization, or mass loss of organic material, and sometimes a combination of these methods. Several

| Site | Litter type | Annual average temp °C | Annual average decomp % | k yr ⁻¹ |
|----------------------------|------------------------|------------------------|-------------------------|--------------------|
| Devon Isl. N.W.Territories | Carex | -13°C | 12-19 | 0.12-0.21 |
| Barrow, Alaska | Dupontia fischeri | -5°C | 15-18 | 0.16-0.20 |
| Finland | Mixed deciduous-spruce | 4.5°C | 30 | 0.36 |
| Finland | Mixed deciduous-spruce | 5.5°C | 42-46 | 0.54-0.62 |
| Bangor, Maine | Coniferous | 6.8°C | 42 | 0.544 |
| Zutphen, Netherland | Deciduous mull-mor | 9.5°C | 54-62 | 0.77-0.97 |
| Zutphen, Netherland | Deciduous mull-mor | 10°C | 69-84 | 1.17-1.83 |
| Coweeta, North Carolina | Mixed forest | 13°C | 50 | 0.70 |
| Athens, Georgia | White oak-dogwood | 15°C | 40 | 0.75 |
| Oak Ridge, Tennessee | White oak | 15°C | 72 | 1.27 |
| Kade, Ghana | Moist semi-deciduous | 26°C | 300 | 15.15 |
| Ibadan, Nigeria | Dry semi-deciduous | 26°C | 300 | 15.15 |
| Nigeria, Ivory Coast | Moist evergreen | 26-30°C | 350-400 | 15.2-20.00 |

Table 7.4: Display of basic reference behaviour, taken from the literature. Litter decomposition rates, annual averages, at different locations in the world. k is the rate coefficient according to first order rate equation, no consideration of pH, soil moisture or substrate quality was made. The data was taken from the literature (Chen et al., 1983, Tietma et al., 1991, Edwards 1975). The annual average temperatures were taken from the National Geographic Atlas of the World (1963).

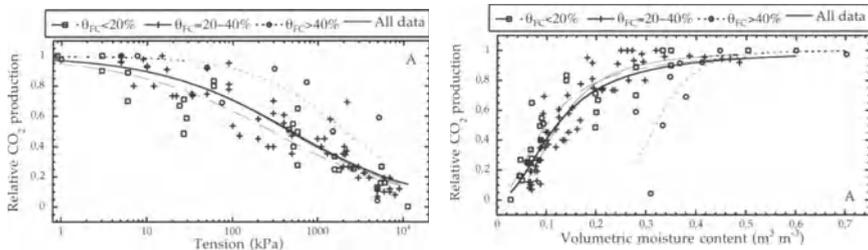


Figure 7.19: Relative CO_2 production as a function of different soil moisture indices. The soils have been divided into three groups depending on soil water retention properties.

authors have compared experiments on decomposition rates (Sommers et al 1980, Skopp et al 1990; Rixon and Bridge 1968; Walse et al 1998). It is difficult to compare the results of these investigations, however, since they used different moisture indices. Apparent inconsistencies of results are due to the different investigative goals, and interpretation of results are difficult because many published reports do not include all information needed to compare the different moisture indices.

The data used was taken from literature, representing more than 30 different types of soils. Most soils are forest soils, but some agricultural soils are also included. The experimental design of the reported studies vary. Most were performed in the laboratory, with disturbance of the soils, although Andren et al.(192) studied mass loss of organic matter through experimental manipulations in the field. Eight experiments measured N mineralization, four experiments measured CO_2 production, and two experiments measured both CO_2 production and N mineralization. N mineralization was measured as the net accumulation of nitrate and ammonia in the soil over the experimental period. CO_2 production was measured either as the accumulated CO_2 production over the experimental period, or as the rate of CO_2 production at certain time

| θ_i | All soils n=107 | | | $\theta_{FC} < 20\%$ n=32 | | | $\theta_{FC} 20-40\%$ n=65 | | | $\theta_{FC} > 40\%$ n=10 | | |
|-----------------|--------------------|------|-------|------------------------------|------|-------|-------------------------------|------|-------|------------------------------|------|-------|
| | K | m | r^2 | K | m | r^2 | K | m | r^2 | K | m | r^2 |
| Ψ | 31 | -0.6 | 0.83 | 19 | -0.5 | 0.89 | 33 | -0.6 | 0.87 | 409 | -0.8 | 0.68 |
| θ_{RFC} | 3.9 | 2.4 | 0.83 | 3.2 | 1.9 | 0.89 | 4.7 | 2.7 | 0.88 | 9.7 | 8.0 | 0.56 |
| θ_{vol} | 72 | 2.0 | 0.77 | 75 | 1.9 | 0.87 | 252 | 2.7 | 0.88 | 9678 | 8.0 | 0.59 |
| θ_{grav} | 82 | 1.9 | 0.69 | 180 | 1.9 | 0.89 | 367 | 2.7 | 0.88 | 219 | 5.5 | 0.39 |
| θ_{RS} | 11 | 2.1 | 0.70 | 11 | 1.8 | 0.79 | 16 | 2.6 | 0.73 | $4 \cdot 10^8$ | 27 | 0.67 |
| V | 0.09 | -2.5 | 0.45 | $9 \cdot 10^{-11}$ | -23 | 0.76 | 0.04 | -2.2 | 0.67 | $3 \cdot 10^{-3}$ | -6.9 | 0.78 |

Table 7.5: The parameters K and m and the coefficient of correlation (r^2) for the model fit to the CO_2 production data. n denotes the number of data points in each regression.

| θ_i | All soils n=209 | | | $\theta_{FC} < 20\%$ n=52 ^a | | | $\theta_{FC} 20-40\%$ n=134 ^b | | | $\theta_{FC} > 40\%$ n=23 | | |
|-----------------|--------------------|------|-------|---|------|-------|---|------|-------|------------------------------|------|-------|
| | K | m | r^2 | K | m | r^2 | K | m | r^2 | K | m | r^2 |
| Ψ | 44 | -0.6 | 0.66 | 31 | -0.5 | 0.55 | 47 | -0.6 | 0.71 | 306 | -0.9 | 0.77 |
| θ_{RFC} | 5.2 | 2.6 | 0.70 | 4.6 | 1.9 | 0.47 | 6.8 | 3.3 | 0.85 | 4.4 | 4.0 | 0.64 |
| θ_{vol} | | | | 99 | 1.9 | 0.46 | 126 | 2.6 | 0.72 | $2.4 \cdot 10^6$ | 13 | 0.65 |
| θ_{grav} | | | | 140 | 1.7 | 0.42 | 194 | 2.5 | 0.71 | 189 | 5.0 | 0.45 |
| θ_{RS} | | | | 14 | 1.7 | 0.48 | 15 | 2.7 | 0.72 | 21 | 4.5 | 0.41 |
| V | | | | $7 \cdot 10^{-3}$ | -3.8 | 0.46 | 0.1 | -2.0 | 0.32 | $4 \cdot 10^{-4}$ | -7.0 | 0.59 |

Table 7.6: The parameters K and m and the coefficient of determination (r^2) for the model fit to the N mineralization data. n denotes the number of data points in each regression. ^{a,b} $n_{\theta_{RS}}=40$ and 102 respectively.

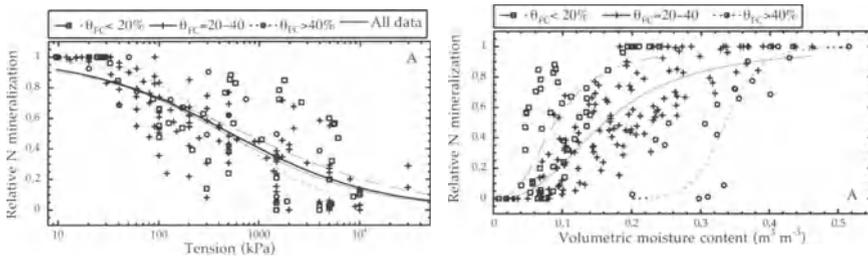


Figure 7.20: Relative N mineralization as a function of different soil moisture indices Ψ (A), θ_{RFC} (B) and θ_{RS} (C). The soils have been divided into three groups depending on soil water retention properties.

intervals. The duration of the experiments varied from 3 days to 3 months, over temperatures of 2 to 40 °C. Some authors used a range of constant temperatures and some used a single constant temperature (Table ??). We tried to find experiments that included a wide range of soil moisture contents, ideally the soil moisture contents corresponding to 0 to -1500 kPa matrix potential (saturation to wilting point) or drier. However, few experiments use the whole range of moisture contents, and we had to settle also for data from experiments that covered only parts of that range. Because not all reports included all the background information needed to perform this study, some background data were estimated from other models or methods of estimation. The most critical estimation was that of soil moisture tension from volumetric soil moisture, and vice versa. This was done for all data sets where either soil moisture tension or volumetric soil moisture were missing. The model used to calculate the moisture-tension curves was presented by Gregerson et al.(1987), and is here called the GHM model. It is based on the log-log linear form of the soil water retention curves in the -5 kPa to -1500 kPa matrix potential range (Brooks and Corey 1964) Soil moisture experiments are often performed on the basis of the water-to-soil mass ratio, or as a function of the soil matrix potential. These

| Reference | Material | Method | Duration | K_w | n_w | N | R^2 |
|------------------------------|--------------|-----------------------|-----------|-------|-------|-----|-------|
| Rosswall and Berg 1972 | Cellulose | Mass loss | 8 weeks | 5.6 | 2.8 | 29 | 0.96 |
| Zöttl 1960 | Humus | N min. | 16 weeks | 5.9 | 3.0 | 3 | 0.92 |
| Mork 1938 | Humus | N min. | 16 weeks | 10.4 | 3.5 | 8 | 0.76 |
| Birch 1958 | Soil | N min. | 5 days | 9.0 | 3.6 | 10 | 0.98 |
| Robinson 1957 | Soil | N min. | 32 days | 3.7 | 3.5 | 15 | 0.77 |
| Theodorou and Bowen 1983 | Soil | N min. | 8 weeks | 3.7 | 2.7 | 6 | 0.77 |
| Campbell and Biederbeck 1975 | Soil | N min. | 2 weeks | 17.5 | 3.8 | 27 | 0.76 |
| Andren et al 1992 | Barley straw | Mass loss | 3 months | 9.1 | 3.8 | 3 | 0.71 |
| Group 1, all data | | | | 9.3 | 3.4 | 101 | 0.92 |
| Tyler 1075 | Soil | CO ₂ prod. | 24 hrs | 1145 | 3.6 | 11 | 0.98 |
| Wildung et al 1975 | Mixed litter | CO ₂ prod. | 24 hrs | 28.4 | 1.5 | 17 | 0.77 |
| Tyler 1974 | Soil | Mass loss | 3 months | 24.6 | 1.4 | 8 | 0.96 |
| Goncalves and Carlyle 1994 | Soil | N min | 2-8 weeks | 16.0 | 1.7 | 30 | 0.87 |
| Group 2, all data | | | | 16.7 | 1.7 | 66 | 0.85 |

Table 7.7: Relationship between relative soil moisture saturation and decomposition rate. K_w and n_w are the coefficients of the Langmuir adsorption isotherm (equation 14). N denotes the number of measurements in each experiment and R^2 denotes the coefficient of determination.

measurements are difficult to relate between soils unless, in the case of soil matrix potentials, you have the hydrological data for the soil in question to use in combination with the decomposition data. Experiments using the relative soil moisture saturation as a parameter, as we have chosen to model it, are less frequent but often very informative. The data can be divided into two groups (figure 7.21). One (group 1), including six nitrogen mineralization experiments and two mass loss experiments over periods of 5 days to 16 weeks shows a slower increase in nitrogen mineralization rate or mass loss rate with relative soil moisture saturation. The other group (group 2) contains two short-term respiration experiments (24 hrs), reported by Tyler (1974,1975) and two longer-term mass-loss experiments (2-8 weeks and 3 months) reported by Wildung et al. (1975) and Tyler (1975). These experiments show a very strong response in decomposition rate in the relative soil moisture saturation interval 0-20% followed by a slight increase in decomposition rate between 20-100% relative soil moisture saturation (figure 7.21). It is probable that the short-term respiration experiments (24 hrs or less) reflect the changes in both the microbial biomass and its activity. A dried-out soil receiving small amounts of water may show a very quick response in carbon dioxide production, but as the carbon dioxide concentration in the soil pores increases, the carbon dioxide production will level off. It is also possible that the short-term experiments reflect the decomposition of a pool of very easily decomposable substrate, unavailable to the microbial biomass at drought conditions. After the addition of water this pool is quickly decomposed. The more long-term experiments would then better represent the integrated soil processes, including inhibitory reactions. The experimental data taken from Tyler (1974) and Goncalves and Carlyle (1994) does not correspond to this theory and no other relevant explanation was found for these results. The medium-term data from nitrogen mineralization and mass loss experiments (group 1) were chosen for the extraction of model parameters. Since the experiments on decomposition of "pure" cellulose and humus give very similar rate regulating functions as experiments on soils, all data in group 1 were used to create a single rate regulating function (i.e. $g_1(\theta) \dots g_4(\theta)$ in equation 1-4 are identical). The parameters K_w and n_w were estimated to 9.3 and 3.4, respectively. All references and parameters extracted from data are listed in table 7.7. The CO₂ production data used in this study showed an increase in CO₂ production over the whole range of soil moisture (Figure 7.19). This may be surprising, because in soils near saturation, decomposition decreases due to oxygen deficiency or slow diffusion of CO₂ (Linn and Doran 1984). One explanation may be that CO₂ production continues as an anaerobic process in many soils near saturation (Rixon and Bridge 1968, Rovira 1953). Another explanation is that experimental conditions used probably differed considerably from the conditions in the field, in that the soil volumes used were small and the

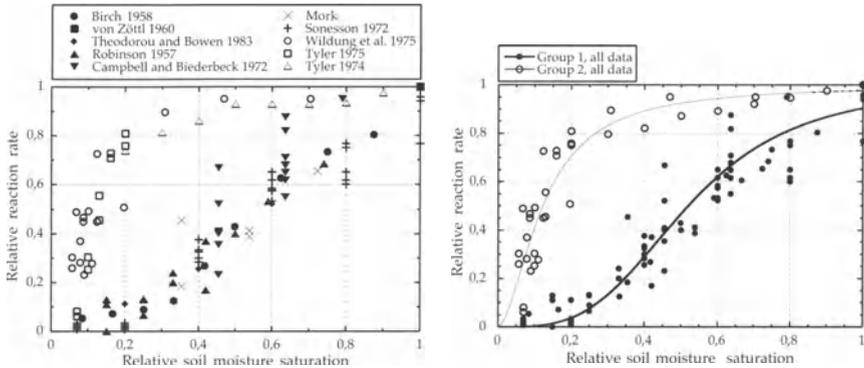


Figure 7.21: Data on decomposition rates at different values of relative soil moisture saturation. The upper figure shows data retrieved from each author, and the lower figure shows the two groups of data. Group 1 contains data from several nitrogen mineralization experiments and a mass loss experiment. Group 2 contains data from two CO_2 emission experiments and a mass loss experiment. The curves show the rate-regulating functions obtained by fitting the model function to the two groups of data.

experiment durations were relatively short, resulting in the oxygen supply being less limiting than in the field. As expected, CO_2 production decreases with decreasing soil moisture. For several of the experiments, however, CO_2 production could be detected in soil with moisture corresponding to Ψ about -8 MPa, or down to a volumetric and gravimetric moisture content of only a few percent. Data are not detailed enough to determine whether there is an abrupt decrease in CO_2 production at a certain dryness. The model could be fitted to all moisture measure data with reasonable accuracy (Figure 7.19 and Table 7.3.5). Over the whole range of soil textures, both matric potential and relative field capacity are moisture indices that yield good model fits to the data ($r^2=0.83$). For the other moisture indices, the accuracy of the model fit then decreases, in the order θ_{vol} , θ_{grav} , θ_{RS} and V . For most soils, one single model function describes the relation between soil moisture and CO_2 production reasonably well. Two soils from the studies of Miller and Johnson (1964) and Andren et al.(1992) differed, however, as they had to be markedly wetter for CO_2 production/mass loss to begin. These two soils, one silt loam and one clay, both show very high water retention capacities ($\theta_{FC} >42\%$). Furthermore, the two soils that show the lowest water retention capacity ($\theta_{FC} <16\%$, from Nyhan (1976) and Quemada and Cabrera (1997) both show maximum CO_2 production at low soil moisture contents. This indicates a trend in the moisture-decomposition relationship with soil water retention capacity. When the soils were divided into three categories according to water retention capacity, a trend in soil moisture-decomposition relation could be detected for some of the moisture indices. All moisture indices gave a difference in behaviour for the two soils with the highest retention capacity compared to the others. For Ψ and θ_{RFC} , this difference was only slight, whereas it was important for all other indices. The soils with low retention capacity did not differ much in behaviour from the middle group of soils when using the indices Ψ , θ_{RFC} , θ_{RS} and θ_{vol} (Figures 7.19). For θ_{grav} , the three groups can be separated, (Figure 7.19), although the soils with lowest water retention capacity still behave fairly similar to the soils with medium retention capacity. When the model was fitted to the data sets from the three groups separately, model fits improved considerably for the moisture indices θ_{vol} , θ_{grav} and θ_{RS} , whereas the separation of data into subcategories had negligible effect on the model fit of Ψ , θ_{RFC} or V (Table 7.3.5). The main difference between the CO_2 production data and the N mineralization data is that nearly all the N experiments showed a decrease in mineral-

ization near saturation. Nitrification is an aerobic process, so when oxygen supply is restricted, it decreases and eventually stops. It appears from the dataset, that maximum mineralization rates occur between -10 and -30 kPa soil matric potential (Figure ??). Near saturation, denitrification often results in losses of gaseous N_2 and N_2O from the soil. These processes are more difficult to examine, requiring more complicated experimental designs. Few of the reports used in this study report denitrification and data reported on net N accumulation in the 0 to -10 kPa range vary greatly. Therefore, nitrogen mineralization under conditions near saturation was not modelled, and data derived from experimental conditions near saturation were excluded from the analysis. The interval for which the derived model functions are valid will also vary with soil type. Therefore, model functions were only derived from the whole range of data in the cases where soil type did not show to be important, i. e. for Ψ and θ_{RFC} . In all other cases, the soils were divided in three groups before model functions were calculated. The interval in which each function is valid can be studied in Figure 7.20. Another important difference between the CO_2 data and the N data is that the N data showed a larger variation, maybe because it comes from a larger number of experiments, covering a greater variety of soils. Another possible reason for the larger variation is that the N mineralization processes are much more specific than CO_2 production processes (N mineralization are carried out by only two microorganisms, whereas C oxidation can be carried out by a range of organisms) and thus may be more susceptible to disturbances. In experiments N mineralization rates were shown to be considerably slower in lab experiments than in the field. Most of the experiments included in this study are laboratory experiments, many including considerable disturbance of the soil such as mixing and drying. Over the whole range of soil textures, the model fit resulted in a weak correlation with data for all moisture indices except Ψ and θ_{RFC} ($r^2=0.66$ and 0.69 , respectively). When model functions were calculated separately for the soils with least and most water retention capacity (as was done for the CO_2 experiments), correlations between model and data for the medium-range soils were acceptable for all indices except V ($r^2=0.71-0.85$). Data from the two groups of soils with low and high water retention capacities scattered more, and yielded correlations in the range of $r^2=0.41-0.77$. The trend in mineralization with soil water retention properties was undetectable for the moisture index Ψ , very slight for θ_{RFC} and increasingly important for θ_{RS} , θ_{vol} and θ_{grav} (Figures 7.20). Consequently, the model fit improved relatively more for these indices when the datasets were divided into subcategories (Table 7.5). Nitrogen mineralization and CO_2 production show very similar trends with changes in soil moisture. In fact, equations are practically interchangeable for the coarser ($\theta_{FC} < 20\%$) and finer ($\theta_{FC} > 40\%$) soils. It is possible that, if the CO_2 experiments had included a greater variety of soil in the mid-range (i.e also soils with a water retention capacity of θ_{FC} 30-40%), the equation derived for CO_2 production for that group would also have been similar to the one derived for N mineralization. Therefore, we pooled the CO_2 and N data and derived one single (Ψ , θ_{RFC}) or three (θ_{vol} , θ_{grav}) model functions to be used for both processes. The results are shown in Table 7.3.5. V was not used at this point, since it generally gave poorer model fits than the other indices. θ_{RS} was also excluded, since it requires the same level of background information as does θ_{RFC} , but overall gives poorer correlations with data than θ_{RFC} . The model functions resulting from the pooling of data are limited in range by the nitrogen data, so that these functions are not valid in a more narrow range than the functions based on carbon dioxide data alone.

Two indices of organic matter decomposition, CO_2 production and N mineralization, can be modelled as a function of moisture using the same type of model function in the range dry soil to -10 to -30 kPa. For soils near saturation, N mineralization decreases whereas CO_2 production stays constant for the soils in this study. If the modelling objective is to find a single moisture-decomposition rate model function valid for all soil types, it is necessary to include some measurements of water potential. The most reliable such function would be based on the full water potential-soil moisture curve, or on relative field capacity. If the objective is to find one or a few model functions that are as little data-demanding as possible, the best option is to model decomposition as a function of gravimetric soil moisture. The similarity of the term for the two processes indicate that the limiting mechanism due to moisture is physical adsorption

of water to organic matter in the soil, and that the adsorption properties of dead humus and live roots are quite similar. Once adsorbed at the surface, plant physiological mechanisms will transfer water into the root, probably proportionally to abundance at the root surface. The function was parameterized by Walse et al. (1998) and Walse and McColl (1999) and the effect on the decomposition rate is shown in Fig. 7.23. The first study showed some ambiguity in the data, but the later study used data of better accuracy and the apparent uncertainty was much resolved. It appears that all types of organic material is exposed to the same effect, suggesting that the water effect is a physical effect on soil particles, regardless of internal particle composition.

pH

Data on the influence of pH on different soil microorganisms and the mass loss of organic material is quite abundant, especially concerning the growth rate of fungi. Results reported in the literature cover a relatively wide pH interval, showing differences in sensitivity between species. When the valence unspecific equation was applied to data and its performance compared to the performance of an empirical equation, the results showed that the valence unspecific equation can be used to model the system behaviour with reasonable accuracy (table 7.12). From the data reported, (table 7.12, figure 7.22), three rate-regulating functions can be extracted: (i) for the impact of pH on fungal growth rate, (ii) for the impact of pH on bacterial growth rate, and (iii) for the impact of pH on decomposition rate. According to the results, bacterial growth rate is more sensitive to low pH than that of fungi. The results from decomposition experiments reflect the integrated process. This response curve lies in between the ones from pure cultures of fungi and bacteria (figure 7.22). Following the hypothesis that soil bacteria are mainly responsible for the decomposition of the more easily decomposed material, while fungi play the main role in the decomposition of lignin and humified material, the response functions are applied as follows:

- Bacterial growth rate is used to model the decomposition of the easily decomposable material.
- Mass loss rates of mixed organic material is used to model decomposition of the holocellulose pool of material.
- Fungal growth rate is used to model the decomposition of lignin and resistant material.

If this hypothesis can not be accepted, the intermediate function ($\phi_2(\text{pH})$) should be used for all pools. In table 7.3.5, the selected model parameters are given.

Aluminum

Data on the combined impact of Al^{3+} and H^+ on soil microbial biomass is sparse and no reported experiments on decomposition rates in the form of mass loss, nitrogen mineralization or carbon dioxide production were found. Apparently, most experimental work has been directed towards the toxicity of aluminum to plant roots, less often towards the toxicity of aluminum to soil microorganisms and very rarely towards the effect of aluminum on integrated soil processes such as decomposition. The analysis was therefore based on experiments on bacterial and fungal growth rates. In the fungal growth experiments, a large number of fungal species have been studied. The exact response in growth rate to a certain $\text{BC}^{2+}/(\text{Al}^{3+} + 3\text{H}^+)$ value varies between species, but a distinct response was found in all cases. One experiment, by Entry et al. (1987), showed a wide variation in response and gave a very poor correlation with all ion exchange functions applied. It was therefore excluded from the parameter determination procedure. Two experiments on the impact of aluminum on bacterial growth were reported in the literature. Unfortunately, the results in the two experiments do not correspond.

| Reference | Material | Exp. cond. | Duration | k (yr ⁻¹) |
|---------------------------------|---|-------------|----------|-----------------------|
| Berg <i>et al.</i> 1991a, 1991b | Leachable | Field 1-7°C | 0.2 yr | 3.0-7.0 |
| Berg <i>et al.</i> 1991a, 1991b | Sugars + amino acids | Field 1-7°C | 3 yrs | 0.3-0.5 |
| Berg <i>et al.</i> 1991a, 1991b | Fats + waxes | Field 1-7°C | 3 yrs | 0.20-0.45 |
| Berg <i>et al.</i> 1991a, 1991b | Cellulose | Field 3.8°C | 3 yrs | 0.35-0.37 |
| Berg <i>et al.</i> 1991a, 1991b | Lignin | Field 1-7°C | 3 yrs | 0.04-0.2 |
| Bunell <i>et al.</i> 1977a | Cellulose | Field 1°C | 3 yrs | 0.25 |
| Bunell <i>et al.</i> 1977a | Fats and waxes | Field 1°C | 3 yrs | 0.69 |
| Mindermann 1968 | Sugars (incl.leachable) | Field | 1 yr | 4.6 |
| Mindermann 1968 | Hemicellulose | Field | 1 yr | 2.3 |
| Mindermann 1968 | Cellulose | Field | 1 yr | 1.4 |
| Mindermann 1968 | Lignin | Field | 1 yr | 0.66 |
| Mindermann 1968 | Waxes | Field | 1 yr | 0.28 |
| Mindermann 1968 | Phenols | Field | 1 yr | 0.13 |
| Tenney and Waksman 1929 | Leachable | Lab 25-28°C | 0.2 yr | 3.4-9.2 |
| Tenney and Waksman 1929 | Sugars+amino acids | Lab 25-28°C | 1.1 yrs | 0.34-1.1 |
| Tenney and Waksman 1929 | Starch+ pectin+ tannin + ureic acids | Lab 25-28°C | 1.1 yrs | 0.67-0.84 |
| Tenney and Waksman 1929 | Cellulose | Lab 25-28°C | 1.1 yrs | 0.77-3.32 |
| Tenney and Waksman 1929 | Hemicellulose | Lab 25-28°C | 1.1 yrs | 0.63-2.07 |
| Tenney and Waksman 1929 | Fats and waxes | Lab 25-28°C | 1.1 yrs | 1.3-3.2 |
| Tenney and Waksman 1929 | Lignin | Lab 25-28°C | 1.1 yrs | 0.13-0.23 |
| Tenney and Waksman 1929 | Crude proteins | Lab 25-28°C | 1.1 yrs | 0-0.25 |
| Stanford <i>et al.</i> 1973 | Forest floor, 11 sites | Lab 5°C | 0.5 yr | 0.26-0.78 |

Table 7.8: Rate constants for the decomposition of substances or pools of substances in organic material at experimental conditions.

| θ_i | All soils n=293 | | | $\theta_{FC} < 20\%$ n=74 | | | $\theta_{FC} 20-40\%$ n=188 | | | $\theta_{FC} > 40\%$ n=31 | | |
|-----------------|--------------------|------|----------------|------------------------------|-----|----------------|--------------------------------|-----|----------------|------------------------------|------|----------------|
| | K | m | r ² | K | m | r ² | K | m | r ² | K | m | r ² |
| Ψ | 39 | -0.6 | 0.72 | | | | | | | | | |
| θ_{RFC} | 4.8 | 2.5 | 0.74 | | | | | | | | | |
| θ_{vol} | | | | 96 | 1.9 | 0.60 | 112 | 2.5 | 0.74 | 4.4·10 ⁵ | 11.8 | 0.64 |
| θ_{grav} | | | | 172 | 1.8 | 0.58 | 220 | 2.5 | 0.76 | 141 | 4.7 | 0.45 |

Table 7.9: The parameters K and m and the coefficient of determination (r^2) for the model fit to the combined CO_2 production and N mineralization data. n denotes the number of data points in each regression.

| Model function | Process | Mechanism | K | R ² |
|----------------------|------------------|--------------------|-----------------------|----------------|
| ϕ_1 (pH) | Bacterial growth | Valence unspecific | 65600 | 0.80 |
| ϕ_2 (pH) | Decomposition | Valence unspecific | 20500 | 0.71 |
| ϕ_{3-4} (pH) | Fungal growth | Valence unspecific | 1050 | 0.80 |
| ϕ_1 (pH,Al) | Bacterial growth | Vanselow | 6.97 10 ⁻² | 0.64 |
| ϕ_2 (pH,Al) | Decomposition | Vanselow | 6.7 10 ⁻³ | 0.59 |
| ϕ_{3-4} (pH,Al) | Fungal growth | Vanselow | 6.7 10 ⁻⁴ | 0.59 |

Table 7.10: Selected parameters for the modeling of the impact of pH on the rate of various soil processes. The functions for bacterial growth and fungal growth are suggested to be used as the functions for decomposition of the easily decomposable material and the lignins, respectively.

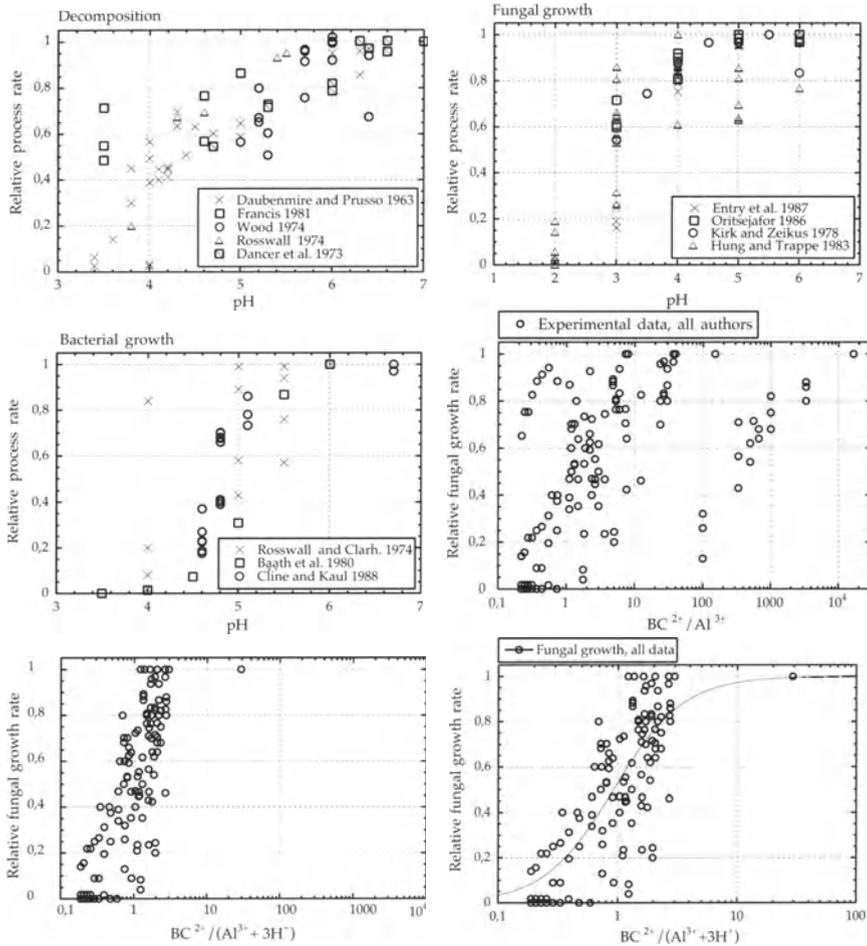


Figure 7.22: Data on rates of decomposition and microbial biomass growth at different pH values. The upper panel shows the data from bacterial growth experiments, the middle panel shows the data from mass loss, carbon dioxide evolution and nitrogen mineralization experiments, and the bottom panel shows the data from the fungal growth experiments. Origin of data is indicated. Data from all experiments (listed in table 7.11) as dependent on the BC^{2+} to $(Al^{3+} + p \cdot H^+)$ molar ratio, at different p values (equation 7.63). The lower panel shows the same data along with the fitted rate-regulating function obtained by using the Vanselow equation (equation 15 and 7.11).

| Reference | ? | N | Best fit p=3 | | | Best fit p=1 | | | Vans. R ² | Unspec. R ² | Gap. R ² |
|--------------------------------|---|-----|-----------------|-----|----------------|-----------------|------|----------------|-------------------------|---------------------------|------------------------|
| | | | n | m | R ² | n | m | R ² | | | |
| Ohno et al 1988 | F | 6 | 3.1 | 2.1 | 0.85 | 1.3 | 1.7 | 0.84 | 0.84 | 0.75 | 0.36 |
| Thompson and Medve 1984 | F | 31 | 3.2 | 3.0 | 0.88 | 1.6 | 2.0 | 0.90 | 0.84 | 0.76 | 0.34 |
| Entry et al 1987 | F | 27 | 2.1 | 0.7 | 0.28 | 0.68 | 0.26 | 0.26 | 0.14 | 0.01 | 0.13 |
| Browning and Hutchinson 1991 | F | 47 | 2.6 | 1.5 | 0.5 | 1.2 | 1.1 | 0.43 | 0.32 | 0.42 | 0.23 |
| Jongbloed, Borst-Pauwells 1992 | F | 21 | 2.7 | 2.4 | 0.85 | 4.4 | 15 | 0.57 | 0.84 | 0.05 | 0.01 |
| Soedarjo and Habte 1993 | F | 10 | 2.7 | 3.1 | 0.79 | 1.6 | 2.6 | 0.87 | 0.76 | 0.75 | 0.39 |
| Fungal growth, all data | | 142 | | | | | | | 0.59 | 0.55 | 0.36 |
| Cline and Kaul 1988 | B | 23 | 2.4 | 1.7 | 0.65 | 1.0 | 1.3 | 0.42 | 0.64 | 0.41 | 0.22 |
| Ohno et al 1988 | B | 6 | 4.0 | 3.5 | 0.97 | 2.2 | 3.4 | 0.96 | 0.80 | 0.66 | 0.27 |

Table 7.11: Combined effect of pH and aluminum concentration on decomposition. Comparison of the n and m parameters in the ion-exchange relation giving the best correlation between data and equation for p=1 and p=3 with the three theoretical ion-exchange equations. F denotes fungal growth, B denotes bacterial growth, N denotes the number of measurements in each experiment and R denotes the coefficient of determination.

| Author | Process | N | Best fit | | Valence unspec. | |
|----------------------------|--------------------------|----|----------|----------------|-----------------|----------------|
| | | | m | R ² | m | R ² |
| Rosswall and Clarholm 1974 | Bacterial growth | 13 | 0.8 | 0.59 | 1.0 | 0.57 |
| Bååth et al 1980 | Bacterial growth | 7 | 1.4 | 0.98 | 1.0 | 0.93 |
| Cline and Kaul 1988 | Bacterial growth | 23 | 2.6 | 0.86 | 1.0 | 0.73 |
| Bacterial growth, all data | | 43 | 1.4 | 0.82 | 1.0 | 0.80 |
| Daubenmire and Prusso 1963 | Mass loss litter | 26 | 0.8 | 0.76 | 1.0 | 0.73 |
| Francis 1982 | CO ₂ ev. soil | 14 | 0.4 | 0.78 | 1.0 | 0.36 |
| Wood 1974 | Mass loss eucalyptus | 19 | 1.05 | 0.54 | 1.0 | 0.54 |
| Rosswall 1974 | Mass loss cellulose | 5 | 1.2 | 0.95 | 1.0 | 0.94 |
| Dancer et al 1973 | N min. | 8 | 0.64 | 0.86 | 1.0 | 0.73 |
| Decomposition, all data | | 72 | 0.62 | 0.79 | 1.0 | 0.71 |
| Entry et al 1987 | Fungal growth | 9 | 1.2 | 0.99 | 1.0 | 0.98 |
| Oritsejafor 1986 | Fungal growth | 18 | 0.6 | 0.93 | 1.0 | 0.87 |
| Kirk and Zeikus 1978 | Fungal growth | 7 | 0.8 | 0.83 | 1.0 | 0.80 |
| Hung and Trappe 1983 | Fungal growth | 60 | 0.8 | 0.81 | 1.0 | 0.80 |
| Fungal growth, all data | | 94 | 0.82 | 0.81 | 1.0 | 0.80 |

Table 7.12: Application of the valence unspecific ion-exchange mechanism and an empirical expression to data for the relationship between pH and decomposition (equation 7.62). N denotes the number of measurements in each experiment and R² denotes the coefficient of determination.

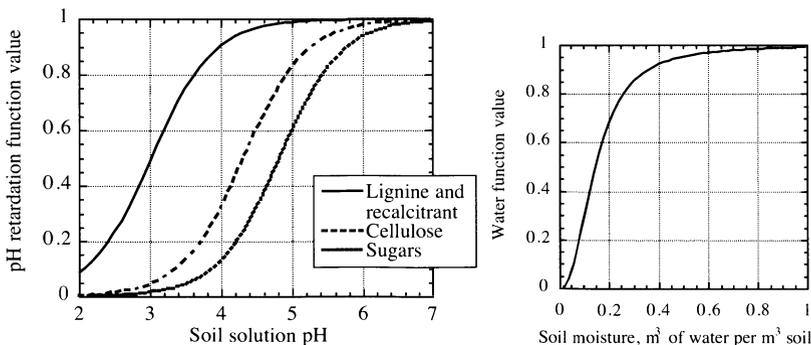


Figure 7.23: The effect of soil solution pH and water on decomposition as used in the computerized model.

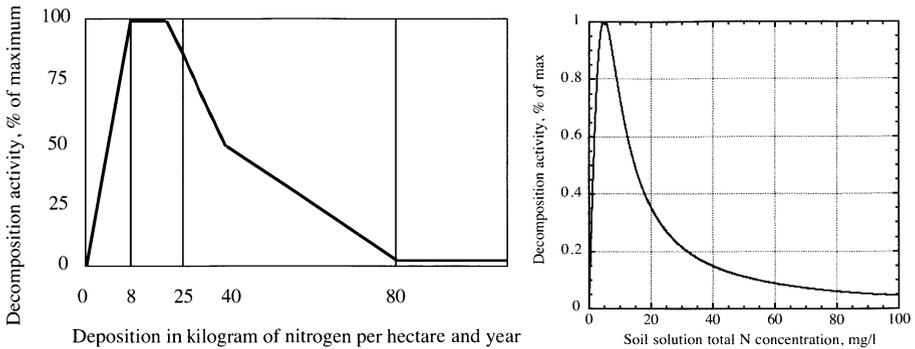


Figure 7.24: The approximate effect of soil nitrogen on decomposition activity as guessed from semi-quantitative information. The effect of the soil nitrogen function on decomposition activity.

Ohno et al. (1988) studied a mixed bacterial culture extracted from soil, and showed a bacterial response very similar to the response found for fungi. Cline and Caul (1988) studied *Bradyrhizobium japonicum*. This experiment showed a much stronger response; the growth rate fell below 50% of the optimal growth rate already at a $BC^{2+}/(Al^{3+}+3H^+)$ value of 10. This result corresponds closer to the results in the pH experiments, where bacteria were shown to be more sensitive to high concentrations of H^+ than fungi. It is difficult to draw any conclusions on the basis of such ambiguous evidence, but since the experiment by Cline and Caul (1988) includes a larger number of measurements, it is favoured in this study (Tab. 7.11, Fig. 7.22). The results show that the n and m factors yielding the best possible correlation were in the range of the ones in the theoretical ion exchange functions. It also showed that a p factor of 1 or 3 gave a substantially better correlation than any other, with a p factor of 3 giving the best correlation. When the three theoretical ion exchange functions were applied to the data from each author and the correlations between data and function were evaluated (table 7.11), it showed that the Vanselow equation describes the relation more closely than the other equations. Applying the Vanselow equation gave coefficients of determination close to those determined from the empirical equations. The valence unspecific equation was also fitted to data with reasonable results, while the Gapon equation gave a very weak correlation with data (table 7.11). Model parameters were found by grouping the experimental data according to process (bacterial growth or fungal growth) and the model parameters were calculated from each group of data (table 7.10). A third model function, the $\phi_2(pH,Al)$ function, was calculated as the intermediate of the functions for bacterial growth ($\phi_1(pH,Al)$) and fungal growth ($\phi_3(pH,Al)$). This estimated model function can be used to model the relation between total decomposition/holocellulose decomposition and $BC^{2+}/(Al^{3+}+3H^+)$ values in parallel with function $\phi_2(pH)$ (table 7.3.5). The calculated model parameter values are listed in table 7.10.

Nitrogen

High N concentration in the soil solution or high N concentration in the substrate can be shown to inhibit decomposition (Berg et al., 1991). One possible mechanism may be product inhibition of the decomposition process. Alternatively, microorganisms may take up nitrogen directly from the solution under high N concentration, and this will be energetically more favourable than deriving it from decomposition. This hypothesis has been displayed in Fig. 7.25 to show the interactions and feedbacks. In both cases, the retardation will respond to the soil solution N concentration. However, a high N content in the substrate will cause a higher release of N per unit weight biomass decomposed. If decomposition is N demand driven, then less biomass can

be decomposed to satisfy the demand when a high N substrate is utilized. In both cases the general expression for product inhibition can be utilized:

$$z(N) = k_N \cdot [N] \cdot \frac{1}{1 + k_R \cdot [N]^z} \quad (7.66)$$

The coefficient and process order must be determined experimentally. There are some circumstantial data that indicate what the coefficients and exponents of this submodel might be. In British Columbia in Western Canada, the deposition of nitrogen is very low (approximately 2 kg nitrogen per hectare and year) and in principle decomposition is retarded by limited external nitrogen access. Small additions of nitrogen under such conditions is known to increase decomposition. In Sweden, the deposition lie in the range from 10 to 25 kg nitrogen per hectare and year. Under such circumstances, more nitrogen rather appear to retard decomposition than promote it. We suggest that the promoting effect continues to approximately 8 kg nitrogen per hectare and year and that the decline starts to become significant above 25 kg nitrogen per hectare and year (a reduction in 20% maybe). In the Netherlands, the deposition of nitrogen is very high to extremely high (40 to 100 kg nitrogen per hectare and year) and under such conditions, decomposition appear to be heavily retarded (at 80 kg nitrogen per hectare and years maybe 90% reduction in activity). In Fig. 7.24 we have tried to show this. The coefficients can be estimated by guessing precipitation and percolation volumes for the examples used and back-calculation of the soil solution nitrogen concentration. Assuming a precipitation of 800 mm and a percolation of 450 mm, we arrive at the values given in Tab. 1. The values are very approximate, and should be subject to proper determination on good field data in the future. The effect of the parameterized expression given above and included in the model is shown in Fig. 7.24.

Heavy metals

It is apparent that some heavy metals such as Cu, Ni or Zn may be so abundant in the soil under polluted conditions, as well as Fe and Mn under natural conditions, that they can compete with base cations at the cell membranes the same way Al would do, the "outnumbering effect" of Al (Sverdrup et al., 1991, 1993). This effect is regarded as separate from toxic effects. Some studies have been published on the toxicity of Al and heavy metals to plants (Tamm and Wiklander 1980, Tyler 1983, Fishner et al 1983). The data are generally derived from laboratory experiments, often on seedlings, as few in situ field measurement are available. We assume the effects of Al and heavy metals to be separable and that the Al effect is not mechanistically coupled to the heavy metal concentration, but that the heavy metal effect is interfered upon by the major constituents H^+ , BC^{2+} and Al^{3+} under acidic soil conditions. Under such circumstances, we can derive a retarding function with the same basic principles. To simplify the problem, it is assumed that the heavy metal occur as divalent ion in its biochemically active form in relation to nutrient uptake and metabolism. On basis of this we can use the

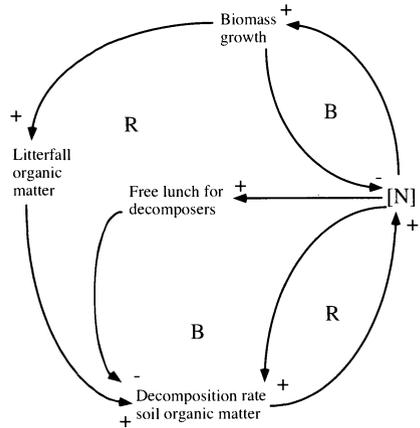


Figure 7.25: *The mental model for the effect of soil nitrogen on decomposition activity displayed as a causal loop diagram. The principles displayed are not fully established, but rather to be seen as a hypothesis for which there is only circumstantial evidence. This principle has been applied in our model.*

| Nitrogen process | k | z |
|------------------|---------------------|------|
| N-Promotion | 0.36 | - |
| N-Retardation | 0.02 | 2.3 |
| Metal | k_M | y |
| Cu | $2.8 \cdot 10^{-7}$ | 0.87 |
| methyl-Hg | $3.2 \cdot 10^{15}$ | 1.75 |
| Pb | $3.8 \cdot 10^3$ | 0.52 |
| Cd | $3.5 \cdot 10^9$ | 0.5 |

Table 7.13: The upper part of the table show the parameter values in the nitrogen promotion-retardation expression. The expression is based on nitrogen concentration in the soil solution in mg/l. The lower part of the table show the retardation coefficients for some metals based on metal concentration in kmol/m³.

Gapon ion exchange selectivity expressions for the adsorption at the bacteria and fungi cell wall (Sverdrup, Warfvinge and Rosen, 1991), and assume Ca, Al, H and heavy metal to be the only occupants of these adsorption sites. The hypothesis that BC-uptake is proportional to the adsorbed fraction base cation at the cell wall is used to yield the expression for the competitive effect of high heavy metal concentration on base cation uptake. If heavy metals are present in outnumbering concentration and no toxic metals are present, and the BC and Al concentration constant, the expression may be simplified:

$$v(M) = \frac{1}{1 + K_M \cdot [M]^y} \quad (7.67)$$

K_M represent the selectivity of heavy metal to the microorganism. This expression was used in parameterization concerning heavy metals. It may be hypothesized that their action is through metabolism interference through enzyme reaction poisoning in the cytoplasm and of extracellular enzymatic activity of the bacteria and fungi. The poisoning effect on enzymes is proportional to the heavy metal concentration, and for bacteria and fungi cytoplasmic effects dependent on the uptake of the heavy metals. The adsorption sites for uptake to the bacteria are subject to competition from divalent base cations and the base cation competitions is negatively affected by H⁺ ions in acid soils. Accordingly, heavy metals cannot disrupt base cation uptake by competition, but base cations, Al or H⁺ ions may disrupt or promote heavy metal uptake. The suggested reaction mechanism is combined with the hypothesis that heavy metal adsorption to the microorganism surface, follow an absorption process similar to ion exchange. Tyler (1983) has presented data that allow a parameterization of the model. Studies on trees indicate that methyl-Hg has a toxic effect on tree root growth, possibly there would be a similar effect on biomass decomposition.

Rate constants

Rate constants for the decomposition of mixed organic materials can be calculated from a large number of field and laboratory experiments (table 7.8). Reports on experiments with separate substrates or groups of substrates such as celluloses and lignins are also relatively abundant. In order to calculate the potential rate constants as defined in this study, however, it is necessary to know the experimental parameters such as temperature, moisture conditions and pH. These parameters are seldom fully reported in the literature. In table 7.8 we have

| Site | Exp. cond. | | | Pool | k yr ⁻¹ | f(T) | g(θ) | ϕ (pH) | k_{pot} . yr ⁻¹ |
|-------------|------------|---------------|---------|----------------------|-----------------------|------|---------------|-------------|---------------------------------|
| | T °C | θ - | pH - | | | | | | |
| Manjärv | 1.0 | 0.3 | 4.2 | Leachable | 4.5 | 0.89 | 0.14 | 0.19 | 194 |
| | | | | Sugars + amino acids | 0.30 | 0.78 | 0.14 | 0.43 | 6.6 |
| | | | | Fats and waxes | 0.25 | 0.78 | 0.14 | 0.43 | 5.5 |
| | | | | Lignin | 0.09 | 0.71 | 0.14 | 0.94 | 1.0 |
| | | | | Pine needles, total | 0.28 | 0.78 | 0.14 | 0.43 | 6.1 |
| Norrliden | 1.2 | 0.27 | 4.3 | Leachable | 4.7 | 0.9 | 0.11 | 0.23 | 183 |
| | | | | Sugars + amino acids | 0.48 | 0.79 | 0.11 | 0.49 | 11.2 |
| | | | | Fats and waxes | 0.40 | 0.79 | 0.11 | 0.49 | 9.3 |
| | | | | Lignin | 0.24 | 0.72 | 0.11 | 0.95 | 3.1 |
| | | | | Pine needles, total | 0.27 | 0.79 | 0.11 | 0.49 | 6.4 |
| Harads | 1.3 | 0.25 | 4.1 | Leachable | 3.2 | 0.90 | 0.08 | 0.15 | 288 |
| | | | | Sugar + amino acids | 0.29 | 0.79 | 0.08 | 0.37 | 12.6 |
| | | | | Fats and waxes | 0.21 | 0.79 | 0.08 | 0.37 | 9.1 |
| | | | | Lignin | 0.02 | 0.73 | 0.08 | 0.92 | 0.4 |
| | | | | Pine needles, total | 0.17 | 0.79 | 0.08 | 0.37 | 7.3 |
| Anundberget | 3.5 | 0.23 | 4.3 | Leachable | 1.0 | 0.96 | 0.14 | 0.22 | 36 |
| | | | | Sugars + amino acids | 0.40 | 0.91 | 0.14 | 0.47 | 6.9 |
| | | | | Fats and waxes | 0.42 | 0.91 | 0.14 | 0.47 | 7.2 |
| | | | | Lignin | 0.25 | 0.88 | 0.14 | 0.95 | 2.2 |
| | | | | Pine needles, total | 0.38 | 0.91 | 0.14 | 0.47 | 6.5 |
| Kappsjön | 3.5 | 0.25 | 3.9 | Leachable | 2.2 | 0.96 | 0.08 | 0.10 | 284 |
| | | | | Sugars + amino acids | 0.3 | 0.91 | 0.08 | 0.27 | 15.5 |
| | | | | Fats and waxes | 0.32 | 0.91 | 0.08 | 0.27 | 16.5 |
| | | | | Lignin | 0.18 | 0.88 | 0.08 | 0.88 | 2.9 |
| | | | | Pine needles, total | 0.38 | 0.91 | 0.08 | 0.27 | 20 |
| Jädraås | 3.8 | 0.24 | 4.2 | Leachable | 3.5 | 0.97 | 0.07 | 0.20 | 257 |
| | | | | Sugars + amino acids | 0.42 | 0.93 | 0.07 | 0.44 | 14.7 |
| | | | | Cellulose | 0.37 | 0.93 | 0.07 | 0.44 | 12.9 |
| | | | | Fats and waxes | 0.40 | 0.93 | 0.07 | 0.44 | 14 |
| | | | | Lignin | 0.12 | 0.90 | 0.07 | 0.94 | 2.0 |
| | | | | Pine needles, total | 0.29 | 0.93 | 0.07 | 0.44 | 10.1 |
| Mästocka | 6.8 | 0.3 | 3.9 | Birch leaves, total | 0.23 | 0.93 | 0.07 | 0.44 | 8.0 |
| | | | | Lignin | 0.16 | 1.17 | 0.14 | 0.89 | 1.1 |
| | | | | Pine needles, total | 0.18 | 1.12 | 0.14 | 0.30 | 4.0 |

Table 7.14: Potential rate constants k_{pot} (corresponding to optimal conditions) for the decomposition of substances or pools of substances in organic material. k is the rate constant as found in the experiment. Decomposition rates and average temperatures were taken from Berg *et al.* 1991a,b. pH and soil moisture were taken from (? , ?) and from unpublished data.

listed the rate constants calculated from various experiments, along with as much information on the experimental conditions as was given in each case. In table 7.14, the potential rate constants are listed for a set of field experiments where all experimental conditions were given in the literature, or could be found elsewhere (Lundmark *et al.* unpublished data). These data are all from litterbag experiments. The pH given is the pH in the litter layer or humus layer and the soil moisture saturation given is the average saturation for the whole soil profile. Therefore, the pH and moisture used for the calculations may differ somewhat from the actual conditions in the litterbags. The experiment pools were grouped to correspond to the model pools; the leachable substances represent the easily decomposable pool and the water soluble substances, the celluloses and the fats and waxes represent the holocellulose pool.

The k_{pot_4} was calculated assuming a turnover time of 300 years at field conditions (a soil moisture saturation of 0.3, a pH of 4.0, and an average temperature of 5° C). The preferred values of model parameters can be summarized as shown in table 7.16.

| Material | Bacteria | Fungi |
|-------------------------------|----------|-------|
| Easily decomposable compounds | 100% | 0 |
| Holocellulose | 50% | 50% |
| Lignin | 0 | 100% |
| Resistant compounds | 0 | 100% |

Table 7.15: *Relative importance of the fungal and bacterial decomposition for the decomposition of different pools of material, as used in this study.*

7.3.6 Discussion of data evaluation

The results show that the methodology and model structure are applicable to available experimental data. However, some of the model parameters presented here are extrapolations of results. Complementary experiments would be necessary to verify the assumptions behind these extrapolations. Such experiments would primarily include studies of bacterial growth and of decomposition rates of various substrates under controlled Al^{3+} and H^+ concentrations. The substrates should include lignin, holocellulose, low-molecular carbohydrates and proteins, and mixtures thereof. These experiments should preferably be set up at non-limiting conditions concerning temperature and moisture. Decomposition experiments should be set up as mass loss experiments and proceed for at least 2 weeks. Furthermore, the variability in soil moisture experiments points to some questions regarding the relation between relative soil moisture saturation and decomposition rate. The modeling of decomposition rates for soil acidification scenarios is thus the primary goal of model development.

1. Observability

- Data found in the literature are detailed enough to create a multi-pool decomposition model. This model may contain four different pools of organic material, representing (i) proteins and low-molecular carbohydrates, (ii) holocellulose, (iii) lignin and (iv) resistant compounds.
- Non-linear functions, preferably derived from a mechanistic interpretation, were necessary to obtain a good representation of measured data.

2. Reaction mechanisms

- The effect of temperature on decomposition rate seems to increase with decreasing nutrient concentration of the substrate.
- The decomposition rate increases with relative soil moisture saturation.
- pH in the soil solution has an impact on the growth rate of fungi and bacteria, as well as on the decomposition rate. Bacterial growth rates are more sensitive to low pH values than fungal growth rates.
- The impact of H^+ and Al^{3+} ions on the growth rate of microbial biomass can be described with ion exchange mechanisms.

7.3.7 DECOMP; Integrated decomposition modelling

Model description

The model operates with reservoirs of organic matter and rate equations for their change. These have been formulated as kinetic equations, depending on state variables and modified by boundary conditions. The basic assumptions of the model are as follows:

| Pool | Eq. | k_{pot} yr ⁻¹ | f(T) E_a kJ mol ⁻¹ | g(θ) | | ϕ (pH) | |
|---------------|-----|-------------------------------|---------------------------------------|---------------|------------|---|--------|
| | | | | K_w - | n_w - | K_{pH} kmol ⁻¹ m ³ | m - |
| Easy | 1 | 250 | 18 | 9.4 | 3.4 | 65600 ¹ | 1 |
| Cellulose | 2 | 11 | 33 | 9.4 | 3.4 | 20500 | 1 |
| Lignine | 3 | 1.8 | 50 | 9.4 | 3.4 | 1050 ² | 1 |
| Resistant | 4 | 0.025 | 53 | 9.4 | 3.4 | 1050 ² | 1 |
| sediments | | 0.0013 | 53 | 110 | 2.454 | 1050 | 1 |
| soil water | | 0.025 | 50 | 110 | 2.454 | 20500 | 1 |
| stream water | | 0.025 | 50 | 110 | 2.454 | 20500 | 1 |
| oxidation | | 0.1*k | 65 | 13.8 | 2.5 | - | - |
| sedimentation | | 0.25 | - | - | - | - | - |

Table 7.16: *Summary of selected parameter values decomposition rates of various pools of substrates*¹. Under the assumption that soil bacteria are responsible for the decomposition of low-molecular carbohydrates and proteins². Under the assumption that soil fungi are responsible for the decomposition of lignins and resistant material.

- Decomposition take place through one step involving 4 basic components of all organic matter in the soil. All substrate differences arising from different vegetation types are assumed to be accounted for by the four-component structure of organic matter, where each vegetation type give rise to a unique organic matter composition. It has been shown earlier that the four basic organic matter fractions are (Walse et al., 1998):
 - Easily degradable sugars, starches and waxes
 - Hemicelluloses and celluloses
 - Lignin
 - Recalcitrant compounds believed to consist of polymers of phenolic and quinonic compounds, organic complexes with non-nutrient metals.
- All decomposition rates are first order
- Several environmental factors affect the rate
 - Soil moisture
 - Soil solution pH
 - Temperature
- The effects of the modifying factors are separable
- Direct oxidation with air catalyzed by sunlight is only significant in the upper soil layer
- Dissolved organic carbon both precipitate and decompose from the soil solution and in lakes
- Input is by litterfall and roots on top, by roots in all other layers
- All precipitation occur to the resistant pool

| Model layer | Carbon bound in | inputs (+) | | sinks (-) | | | | | |
|-------------|-----------------|------------|------|-----------|------|-----|------|------|----|
| 1. layer | Easy pool | litter | root | - | - | dec | - | - | ox |
| | Cellulose | litter | root | - | - | dec | - | - | ox |
| | Lignin | litter | root | dec | - | dec | - | - | ox |
| | Recalcitrant | - | prec | dec | - | dec | - | - | ox |
| | Soil water | - | - | dec | flow | dec | prec | flow | ox |
| 2. layer | Easy pool | - | root | - | - | dec | - | - | - |
| | Cellulose | - | root | - | - | dec | - | - | - |
| | Lignin | - | root | dec | - | dec | - | - | - |
| | Recalcitrant | - | prec | dec | - | dec | - | - | - |
| | Soil water | flow | - | dec | flow | dec | prec | flow | - |
| 3. layer | Easy pool | - | root | - | - | dec | - | - | - |
| | Cellulose | - | root | - | - | dec | - | - | - |
| | Lignin | - | root | dec | - | dec | - | - | - |
| | Recalcitrant | - | prec | dec | - | dec | - | - | - |
| | Soil water | flow | - | dec | flow | dec | prec | flow | - |
| 4. layer | Easy pool | - | root | - | - | dec | - | - | - |
| | Cellulose | - | root | - | - | dec | - | - | - |
| | Lignin | - | root | dec | - | dec | - | - | - |
| | Recalcitrant | - | prec | dec | - | dec | - | - | - |
| | Soil water | flow | - | dec | flow | dec | prec | flow | - |
| Lake | Lake water | dep | - | - | flow | dec | prec | flow | - |
| | Sediment | - | - | - | prec | dec | - | - | - |

Table 7.17: *Matrix of sources and sinks in the model system. litter and root are input fluxes from the growth of roots and litterfall to the ground, dec is the contribution from decomposition process, prec is precipitation of DOC from the solution which always go to the recalcitrant pool, ox is the rate of oxidation with air, and finally, flow is the input or output with solute transport. part is partitioning to the fish, based on total lake water concentration.*

The total rate of mass loss will be the sum of the rates of the individual decomposition processes:

$$r_{mass\ loss} = r_{easy} + r_{cellulose} + r_{lignine} + r_{residual} \quad (7.68)$$

Residual organic matter fraction is decomposed slowly, but also created in the decomposition of lignin. A part of the lignin substrate consists of fibers and parts which resist normal decomposition. The effect of different inhibitors and promoters are assumed to be independent functions which may be linearly coupled in the rate expression. Temperature, soil moisture, soil acidity and the presence of heavy metals in the soil solution are factors that must be considered in a decomposition model incorporating environmental and climatic variability or change. k is the decomposition rate coefficient. The effect modifying factors are hypothesized to be multiplicative. For each of the factors affecting the biomass decomposition rate, a specific function can be deduced from the hypothesized mechanism or alternatively from empirical observations. Some of the modifying functions involved in degrading of different organic matter fractions may be common to the decomposing organisms and not to type of organic substrate.

litter and *root* are input fluxes from the growth of roots and litterfall to the ground, *dec* is the contribution from decomposition process, *prec* is precipitation of DOC from the solution which always go to the recalcitrant pool, *ox* is the rate of oxidation with air, and finally, *flow* is the input or output with solute transport. *part* is partitioning to the fish, based on total lake water concentration. The amount bound in the easily degradable pools, cellulose and lignin in

the second, third and fourth soil layer is small as compared to the amount in the recalcitrant, and it was thus omitted. The last pool for mercury, fish, was calculated using equilibrium partitioning with lake total mercury concentration. The whole equation system for mass of each substance pool (easily decomposable, cellulose type material, lignin, slowly decomposable, mercury) in each soil layer is based on equations based on mass balance:

$$\frac{d}{dt}m = in + prod - sinks - out \quad (7.69)$$

and in the aqueous phase of every layer and in the lake;

$$\frac{d}{dt}V \cdot [C] = in + prod - sinks - out \quad (7.70)$$

Of the decomposed amount, 50% is converted to DOC from the easy pool and cellulose, and 40% of the lignin and recalcitrant decomposed. 50% goes to CO₂ and 10% of the decomposed amount of lignin and recalcitrant matter is converted to new recalcitrant matter. The equation system is repeated for 4 layers. The rate of oxidation apply only to the upper layer and then also only to the easily degradable and the cellulose fractions. The oxidation rate is constant and only influenced by water and temperature. A separate rate of oxidation apply to the DOC in the soil solution. In the lake, the sedimentation rate substitutes for the precipitation rate. The sedimented matter decomposes with a very slow rate. The rates of organic matter input is a function of the composition of the incoming litter. This depend on the vegetation type and will change when the species changes occur in the vegetation. In each layer a mass balance for DOC is made:

$$r_{DOC} = r_{decomposition} \cdot X_{DOC} - r_{DOC/decom} - r_{precip} \quad (7.71)$$

Solute flow also move DOC;

$$in - out = Q_{in} \cdot [C]_{in} - Q_{out} \cdot [C] \quad (7.72)$$

r is a rate, usually in kg ha⁻¹yr⁻¹, V is water volume in m³ and Q is water flow in m³ha⁻¹yr⁻¹.

The core of the model is the system of mass balances for the pools of the organic matter fractions, soil solution DOC and DOC in the lake, expressed as ordinary non-linear differential equations. The whole equation system for one soil layer is repeated in every one of the four soil layers in our model:

$$\frac{dEasy}{dt} = r_{inE} - r_{Easy} \quad (7.73)$$

$$\frac{dCellulose}{dt} = r_{inC} - r_{Cell} \quad (7.74)$$

$$\frac{dLignine}{dt} = r_{inL} - r_{Lignine} \quad (7.75)$$

$$\frac{dResistant}{dt} = r_{inR} + 0.1 \cdot (r_{Easy} + r_{Cell} + r_{Lign}) - r_{Resist} + r_{Precip} \quad (7.76)$$

$$\frac{dCO_2}{dt} = 0.4 \cdot (r_{Easy} + r_{Cell}) + 0.5 \cdot (r_{Lign} + r_{Resist}) + r_{DOC} \quad (7.77)$$

$$\frac{d[DOC]}{dt} \cdot V = 0.5 \cdot (r_{Easy} + r_{Cell}) + 0.4 \cdot (r_{Lign} + r_{Resist}) - r_{DOC} Q_{in} \cdot [DOC]_{in} - Q_{out} \cdot [DOC] - r_{Precip} \quad (7.78)$$

r is a rate, usually in kg ha⁻¹yr⁻¹, V is water volume in m³ and Q is water flow in m³ha⁻¹yr⁻¹. The rate of oxidation apply only to the upper layer and then also only to the easily degradable sugars and waxes and the different cellulose fractions.

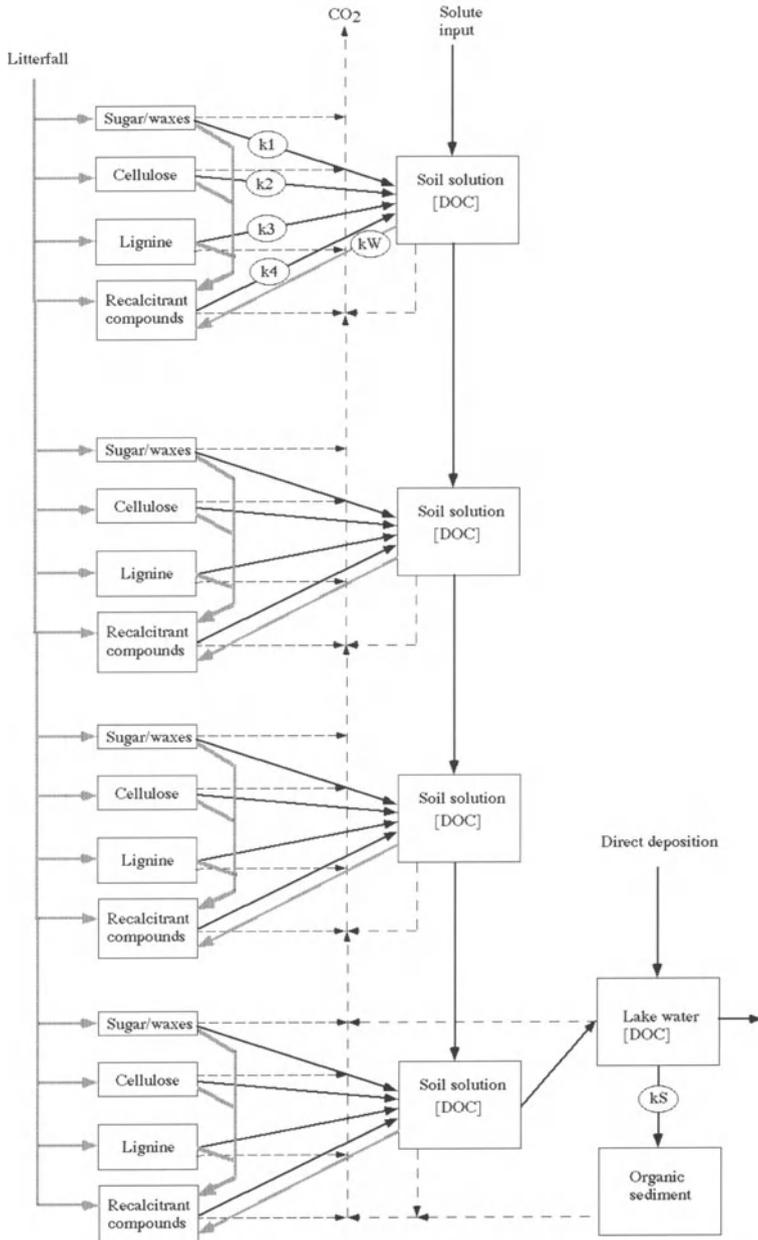


Figure 7.26: Diagram of the STELLA model built for the soil system. The model has four soil layers and a lake.

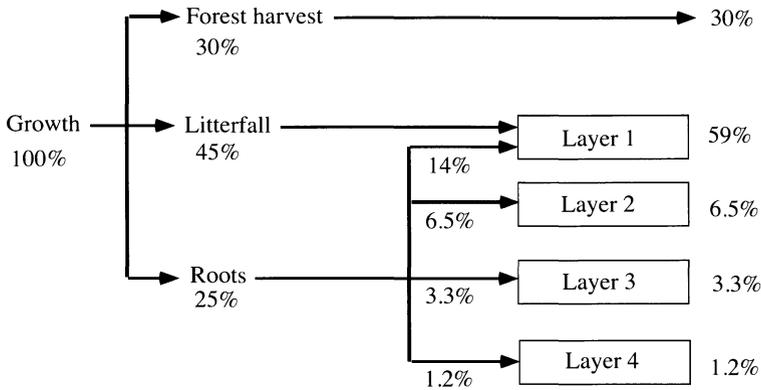


Figure 7.27: Routing of the growth produced to harvest, litterfall and root biomass. The root biomass production has been divided among the different soil layers proportionally to horizontal distribution of root weight. Typical root distributions from the literature was used.

| | Spruce | Pine | Birch | Beech | Oak |
|--------------|--------|------|-------|-------|-----|
| Easy | 10 | 12 | 20 | 20 | 18 |
| Cellulose | 65 | 65 | 65 | 65 | 65 |
| Lignin | 20 | 19 | 14 | 14 | 10 |
| Recalcitrant | 5 | 4 | 1 | 1 | 7 |

Table 7.18: The table show the approximate content in weight % of the different organic matter fractions in litterfall for different tree species. The data was taken from a German study from Solling and a Swedish study.

The total rate of mass loss will be the sum of the rates of the individual decomposition processes. In the lake, the sedimentation rate substitutes for the precipitation rate used in the soil. The sedimented organic matter decomposes with a very slow rate in lake sediments. The rates of organic matter input is a function of the composition of the incoming litter. This depend on the vegetation type and will change when the species changes occur in the vegetation. In the experimental model described here, we operate with four soil layers and an appended lake. Figs. 15.1 and 15.2 show the flowsheet for each individual layer in the model and the whole integrated model. Growth was distributed between soil layers as shown in Fig 7.27. Important for the model is the distribution of total net primary production to stem growth, litterfall and root growth. This has been illustrated in Fig. 7.27. It shows the routing of the growth produced to harvest, litterfall and root biomass. The root biomass production has been divided among the different soil layers proportionally to horizontal distribution of root weight. Typical root distributions from the literature was used. Tab. 14.10 show the approximate root distribution as weight % for different tree species, and the approximate content in weight % of the different organic matter fractions in litterfall for different tree species (Kimmins 1998).

The model does not contain a dynamic soil water model, but is operated with a number of simple algorithms and forcing functions. In the model, water going to the lake was taken from the top and bottom soil layer. During flows lower than the average annual flow rate, 0.8 m yr⁻¹ at Asa, DOC is taken from the lower layer with a flux maximized to that value. All

water volumes above that value is taken from the top soil layer with the DOC concentration in that layer. This cause high flows to have higher DOC concentrations, and low flowrates to be composed of clearer water. This is corresponding to say that water at low flow originates from groundwater and deeper parts of the soil, and that the water table rise and give rise to flow from organic border zones to rivers during periods of high flow. Field reference behaviour demands that DOC should go up during high flow such as in the spring in Swedish forests. The decomposition rate for each of the reactions given above, was assumed to follow a first order kinetic rate law with respect to substrate, modified by functions dependent on water (w), soil chemistry (pH), temperature (T), soil availability of nitrogen (N) and sometimes even heavy metals (M). For each organic matter fraction we have the kinetic rate expression:

$$r_i = k_i \cdot m_i \cdot f(w) \cdot g_i(pH) \cdot j_i(T) \cdot z(N) \cdot v(M) \quad (7.79)$$

where the different organic matter fractions each have their particular kinetic coefficient and modifying functions attached. It is usual to express the rate of microbially mediated reactions by first order kinetics with respect to the substrate, or as a semi-first order Michaelis-Menten kinetic expression, taking into account saturation of the rate with respect to substrate. Residual organic matter fraction is decomposed slowly, but also created in the decomposition of lignin. A part of the lignin substrate consists of fibers and parts which resist normal decomposition. The effect of different inhibitors and promoters are assumed to be independent functions which may be linearly coupled in the rate expression. Temperature, soil moisture, soil acidity and the presence of heavy metals in the soil solution are factors that must be considered in a decomposition model incorporating environmental and climatic variability or change. k is the decomposition rate coefficient. The rate modifying factors are hypothesized to be multiplicative and separable. For each of the factors affecting the biomass decomposition rate, a specific function can be deduced from the hypothesized mechanism or alternatively from empirical observations. The assumption of separability allow us to determine

them in experiments where only one factor is varied at a time. This hypothesis was tested by Walse et al. (1998) and found to be good for this purpose. Some of the modifying functions involved in degrading of different organic matter fractions may be common to the decomposing organisms and for some cases be independent of the type of organic substrate. Direct oxidation with the oxygen of the air also makes a small contribution, and this effect is strongly catalyzed by sunlight. We have at present ignored it in the model. A simplified model based on pH-response experiments was used for the response to soil acidity. The function takes on different values for sugars, celluloses, lignin and recalcitrant organic matter. Organic matter decomposition rate is dependent on the moisture saturation of the soil, and increases with it. The dependence of decomposition on temperature was expressed using an Arrhenius type of expression. Tab. 7.16 show some typical decomposition rate as observed at different parts of the world. The data was taken from the literature. The data can be used for determining rate coefficients. Berg (1985, 1986, 1988, 1989, 1990, 1991) provide a number of experiments from

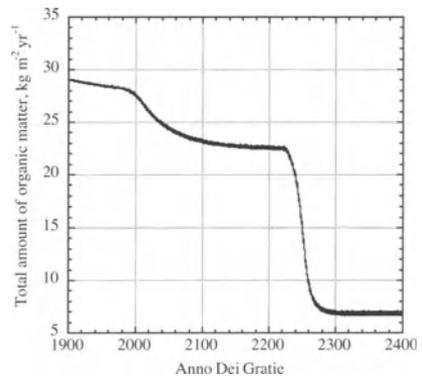


Figure 7.28: Display of the reference behaviour tests, varying from an annual average of 0°C , 8°C and 18°C . The lower figure show the total production of organic matter as input to the soil during the period. The amounts added corresponds to 3, 7 and $20 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$ of forest production. Units are in $\text{kg organic matter per m}^2$ of land.

which decomposition rates for cellulose and lignin can be estimated. He also provide data for estimating rapid ion leaching rates. Study of historical data, and decomposition in the southern side of the Swiss Alps, at Bellinzona, indicate that the rate of decomposition of the residual part is low, 0.0013 yr^{-1} or a half-life of 800-1,000 years (Dr. Peter Blaser, WSL, Schweiz). Annual average temperature at the Bellinzona site is approximately $10\text{-}12^\circ\text{C}$. The Walse-Sverdrup-Berg model for multi-step decomposition was programmed in the STELLA systems analysis development environment. The model was tested against a) conventional reference behaviour as can be derived from generic textbooks, and b) representative data from the coniferous forest located at the sites in the Asa Forest Research Park case study. The model has later been integrated into the SUFOR version of PnET and ForSAFE.

Results

Investigating reference behaviour

A primary result is the construction of this decomposition model that can be used to explore the basic behaviour of the model system. It is a strength of the model that it has been fully parameterized on wholly independent experimental data. It will be a task of the SUFOR programme to test and further develop the model before it is included in the ForSAFE model system. The initial test of the model is to test some well known reference behaviour scenarios:

1. The model should respond to increased temperature with increased rates of decomposition.
2. The model should show larger pools of soil organic matter in cold climate and smaller pools in warm climates.
3. The larger pools in cold climate should contain more material resistant to decomposition than under warm conditions.

In order to test these scenarios, and to be able to see the response more clearly, the soil chemistry was set constant over time. Between the different temperature scenarios, soil water content was held constant and nitrogen cycling was assumed to be complete, thus no nutrient limitation. The growth rate of the vegetation was allowed to increase with temperature, reflecting higher total primary production in warmer climate. Display of the three different temperature regimes used as input to the reference behaviour tests, varying from an annual average of 0°C , 8°C and 18°C is shown in Fig. 7.28. The lower figure show the total production of organic matter as input to the soil during the period.

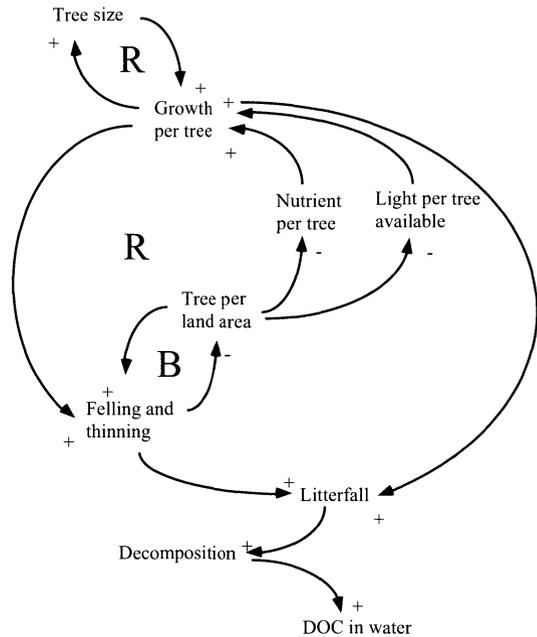


Figure 7.29: Management interferes with the decomposition cycle. Thinning and felling create larger amounts litterfall which become subject to decomposition, and DOC in the percolating water will increase after such events.

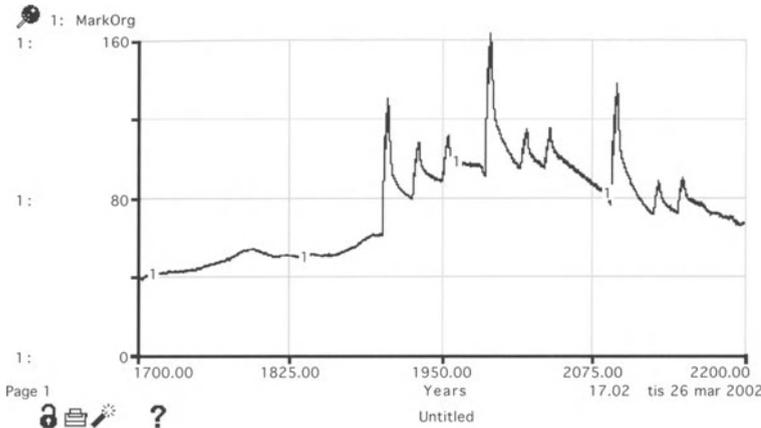


Figure 7.30: The graph shows the accumulation of carbon in the soil with Norway spruce stands since 1700. No consideration is made at this point to take into account the change during 1650-1750 from a broadleaf-dominated landscape to a Norway spruce forest. With the decline after 2050, the growth and litter production rates decline, causing soil reservoirs slowly to decrease.

| Process | Rate | substrate | water | soil chemistry | climate |
|-----------|--------------|--------------|--------|----------------|--------------|
| easy | fast | minimal | strong | strong | small |
| cellulose | intermediate | small | strong | intermediate | intermediate |
| lignin | small | intermediate | strong | small | strong |
| residual | minimal | strong | strong | small | strong |

Table 7.19: A classification of the different factors affecting the overall decomposition rate (process).

Amounts of organic matter in the soil under three different temperature regimes is shown in Fig. 7.28. The increased growth and increased decomposition under increased temperature lead to more easily decomposed and cellulose in the soil. Cold climate seems to enhance the relative amount of recalcitrant organic matter. Warmer climate leads to smaller total amounts of organic matter in the soil. This is in good correlation with conventional wisdom. Units are in kg organic matter per m² of land. The concentration of dissolved organic carbon in the soil solution in the soil under three different temperature regimes corresponded to 20, 40 and 100 mg DOC/l. The soils in warmer climate show higher soil solution DOC concentrations in the top layer, but at 1 meter depth in the soil, the differences are smaller and not so significant. Variation within the year in dissolved organic carbon in a lake under three different temperature regimes for DOC in the downstream lake is shown in Fig. 7.32. The variation during a year of accumulation and release is shown in the diagram at the bottom. The diagram show that organic matter is accumulated during the growth season and partly released again during the fall. For all scenarios, there was a net annual accumulation in the soil. The reference behaviour implies that we have a soil total storage to input rate ratio of 28 years for the cold climate system, 10 years for the temperate climate system and 1 year for the warm climate system.

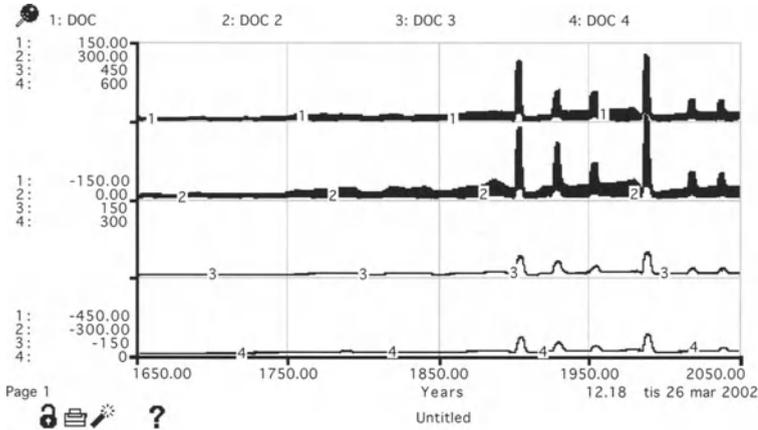


Figure 7.31: The graph shows predicted concentration of dissolved organic matter (DOC) in the layers of the first meter of the soil, $g\ l^{-1}$. The spikes are generated by clear-fellings and thinnings in the stands.

| Climate | Cold | Temperate | Warm | Units |
|-----------------------------------|------|-----------|-------|---------------------|
| Above ground organic matter | 3-10 | 7-20 | 15-40 | $kg\ m^{-2}$ |
| Soil organic matter | 28 | 23 | 7 | $kg\ m^{-2}$ |
| Organic matter input rate | 1 | 2.3 | 7 | $kg\ m^{-2}yr^{-1}$ |
| Soil organic matter turnover time | 30 | 10 | 1 | yr |
| Average annual temperature | -1.5 | 7 | 24 | $^{\circ}C$ |

Table 7.20: A summary of the size of the different organic matter pools and the approximate flux through the system on an annual basis.

Our assessment of the model is that it shows the behaviour that corresponds to the reference behaviour conventional wisdom will demand. The model appear to be very dynamic with a good response to changes in the input parameters. Tab. 7.19 show the relative importance of the different components of the decomposition in soils. If the different factors are added up, it becomes apparent that the easy and cellulose fractions only pay an important role for short term variations, whereas lignin and in particular the residual pool determine the general levels and the really long term dynamics. Fig. 14.2 show the input or organic matter from forest growth and management interventions in the forest system.

Testing at Asa Research Park

For the testing of the model an imaginary site was generated, using generic or average field parameters from the Asa Forest Research Park. The data was generated to be able to do generic dry-runs to investigate the approximate behaviour of the model. The test results from Asa have been shown in Fig. 7.30 and 7.32, they show several interesting features. In the input of organic matter, repeated stand thinnings and clear-cuts with log-only removal was included.

| Process | Value |
|------------------------------------|---|
| Productivity | 9.7 m ³ stemwood ha ⁻¹ yr ⁻¹ |
| Royal Park area | 3,600 ha |
| Forest Research Park area | 1,200 ha |
| Norway Spruce | 60% |
| Scots Pine | 25% |
| Birch | 15% |
| Soil organic matter amount, 0-0.5m | 50-100 kg m ⁻² |
| New organic matter | 1.2-2 kg m ⁻² yr ⁻¹ |
| Soil nitrogen amount, 0-0.5m | 0.3-0.6 kg m ⁻² |
| Nitrogen flux in the soil at 0.5m | 0.015 kg m ⁻² yr ⁻¹ |
| Soil moisture content at 0.3m | 0.2 m ³ water m ⁻³ soil |
| Soil pH, O-layer | 4.3 |
| Soil pH, E-layer | 4.6 |
| Soil pH, B-layer | 5.2 |
| Soil C/N ratio | 25-30 kg/kg |
| Soil temperature | 6°C |
| Acidity deposition | 10-13 kg S ha ⁻¹ yr ⁻¹ |
| Nitrogen deposition | 14-16 kg N ha ⁻¹ yr ⁻¹ |
| Annual precipitation | 8000 m ³ water ha ⁻¹ yr ⁻¹ |
| Annual percolation | 6500 m ³ water ha ⁻¹ yr ⁻¹ |
| Annual runoff | 4500 m ³ water ha ⁻¹ yr ⁻¹ |
| Moose population | 0.15 moose equivalents ha ⁻¹ |
| Moose/deer ratio | 0.2 |

Table 7.21: *Asa Research Park case study. Input data for a hypothetical average site in the Asa Research Park. In the research park, a total of 366 soil sampling sites have been established.*

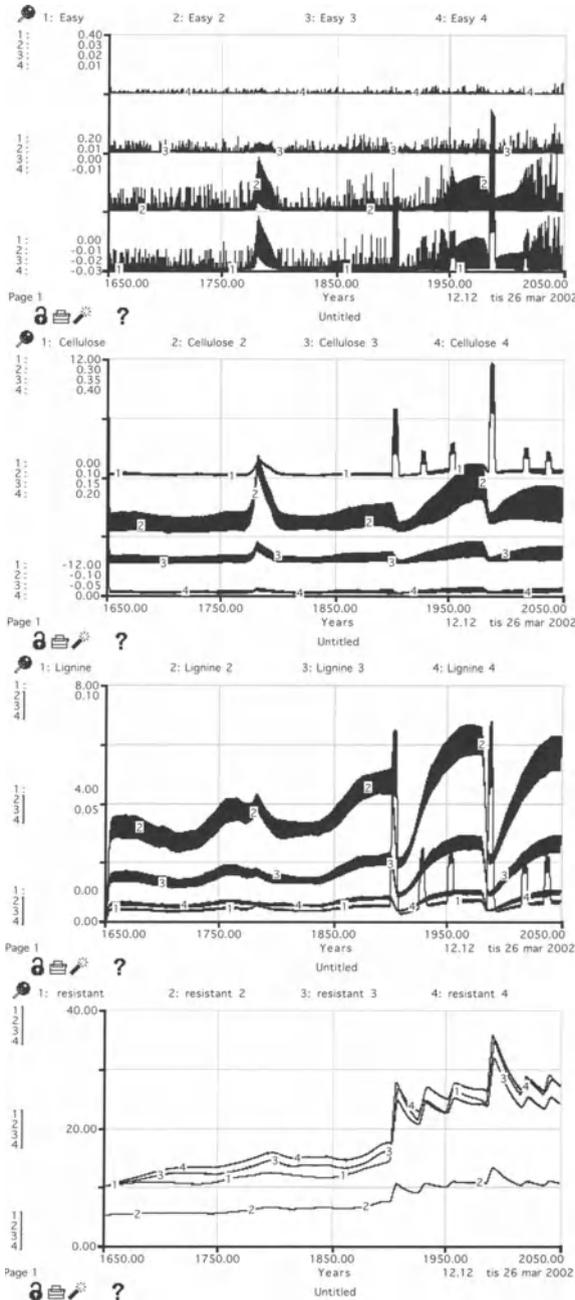


Figure 7.32: Predicted amounts of easily decomposable constituents in kg m^{-2} . The large increase in organic matter during the period 1900 to 2100 is caused by a general increase in growth, caused by increased availability of nitrogen and better management.

The result shows that these management efforts have a large effect on the result in terms of organic matter dynamics and water quality. Leaving the slash from the thinnings and the harvest is very important for maintaining a net accumulation in the soil. If slash removal is practiced, then soil stores of organic matter will start to decline over the long run. This implies that the forest is turned to a net emitter of CO_2 instead of being a net accumulator.

7.4 Nitrogen Immobilization

7.4.1 Introduction

Long term immobilization in forest ecosystems has been estimated using present content divided by soil age. Such estimates ignore the occasional purging of the system when forest fires occur, and also effects of denitrification and fixation. Immobilization as estimated from soil age and stored amount has a value of 0.5-1.5 kg N ha yr. This is very much lower than estimates for present immobilization. At present many forest ecosystems accumulate 7-15 kg/ha yr in the forest floor. The present models are only exploratory in their formulation, not to be confused with a proper process model. They must be seen as a way to approximate a reasonable value.

7.4.2 Theory

The immobilization in Swedish forest soils is driven primarily by retarded decomposition, where nitrogen is locked into recalcitrant organic matter. Uptake by increasing microbiological communities in the soil represent only intermittent phenomena. Fig 7.33 show the process. Three alternatives are at present being evaluated:

1. Bioactivity approach

- Immobilization is caused by incomplete or retarded decomposition and;
- Immobilization is caused by microorganism uptake dependent on available N after uptake to trees
- Abiotic immobilization of ammonium to lignin

2. The immobilization capacity approach

- Immobilization is limited by the immobilization capability which is empirically correlated with the C/N ratio of the soil

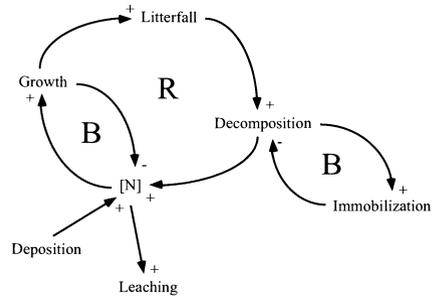


Figure 7.33: Causal loop diagram for the immobilization process.

Decomposition and hence also the fraction not decomposed, is known to be influenced by soil pH, soil water and temperature. The assumptions made in this approach are:

1. Assumptions for the retarded decomposition:

- Immobilization is caused by retarded decomposition of litterfall.
- Retardation is dependent on soil wetness, temperature, pH, and Al.

2. Assumptions for microbiological uptake:

- Immobilization may be caused buildup of microbial biomass.
- Buildup of microbial biomass is dependent on soil solution concentration.

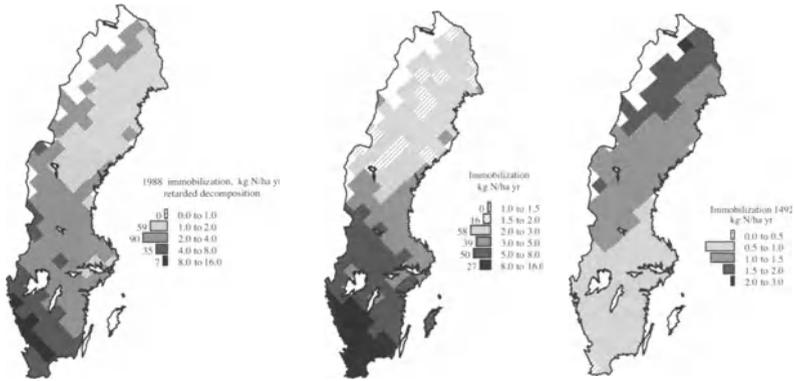


Figure 7.34: Immobilization assuming all immobilization to be based on the N bound in the carbon cycle. Immobilization assuming all immobilization to be microbial action. Historic immobilization, assuming retarded decomposition.

- Microorganism growth and microorganism decomposition is dependent on soil wetness, temperature, pH, and Al.

The C/N approach has been suggested by de Vries (1992). The basic principle is that a part of all N not taken up can be immobilized as long as the C/N ratio of the soil has not reached its minimum value:

$$N_i = (N_{dep} - N_u - N_L) \cdot \left(\frac{C/N - C/N_{min}}{C/N_{max} - C/N_{min}} \right) \quad (7.80)$$

In the expression C/N is the presently observed value. This implies that as the C/N ratio approaches C/N_{min} , then immobilization is reduced to nothing. de Vries (1993) give a value of $C/N_{min}=10$ and $C/N_{max}=50$. This value indicate at which C/N ratio immobilization of all excess N is complete. In soil bacteria, the C/N ratio is 10-12. In warmer climates and in areas with much N deposition, the C/N ratio will tend to go towards lower values, approaching that of soil microorganisms. In spruce needles, $C/N=50-70$, in beech leaves, $C/N=40-50$, and typically the C/N ratio of fresh litter in the forest floor will be close to these values. A soil that has reached a C/N ratio of 10-12 can generally not store more N. The cause for this is that no other forest organism is able to incorporate more N per unit C. There is one important and often occurring exception to this, when there is formation of new soil or a buildup of organic matter. Then N immobilization will simply continue at the rate of organic matter buildup.

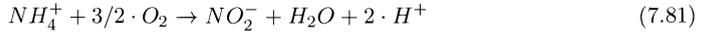
7.4.3 Results

The expression was been calibrated in two ways for the test calculation; (1): Immobilization in northern Sweden was assumed to be 0.5 kg/ha yr at soil pH 5.5 and N deposition 2 kg/ha yr, (2): Immobilization was assumed to be 8 kg/ha yr at present at soil pH 4.2 and the deposition 30 kg/ha yr. The immobilization estimate for present is based on a study on denitrification by Sverdrup and Ineson, (1994). The expression has been tried on the critical loads data base for Sweden. The expressions give higher immobilization in colder areas and areas with very acid soils. For Sweden, the different methods yield similar values for N immobilization.

7.5 Nitrification

7.5.1 Nitrification processes

Nitrification is carried out by bacteria in the soil in a two step mechanism, mediated by two different genera of bacteria, *Nitrosomas* oxidizing NH_4^+ to NO_2^- , and *Nitrobacter* oxidizing nitrite to nitrate. Generally they are either both active or both not active in the soil. Indications has been made that this is correlated to the C/N ratio of the soil. The two overall reactions the bacteria carry out are for *Nitrosomas* the initial oxidation from NH_4^+ to NO_2^- :



and for *Nitrobacter* which carry out the subsequent oxidation of NO_2^- to NO_3^- :



7.5.2 Theory

Generally such biochemical processes as nitrification are modelled by the use of Michaelis-Menten or Monod types of kinetic expressions (Chen et al 1983). Here a Michaelis-Menten equation, modified with the function, f_i to account for the effect of dissolved toxic substances (H ions, Al, heavy metals), the availability of water to the process and the effect of temperature:

$$r_{nitr} = k_{nitr} \cdot \frac{[\text{NH}_4^+]}{K_s + [\text{NH}_4^+]} \cdot \prod_i f_i \quad (7.83)$$

Typical values for the rate coefficient k_{nitr} is given in Tab. 1. K_s is the saturation coefficient, which marks the point where the rate becomes a zero order process, and proceeds at a constant rate. $f(TX)$ is the inhibitory function taking the effect of low pH value and a high concentration of Al or heavy metals into consideration. Attempts have been made to quantify the functions (Russel et al., 1925; Sabey et al., 1959; Tyler, 1974a, 1974b, 1975, 1976a, 1976b, 1977; Tyler and Westman, 1979; Rueling and Tyler, 1979; Tyler et al, 1989; Bossel, Metzler and Schäfer, 1985) but work remains to give the estimates a firm experimental and theoretical basis. The nitrification rate depends mostly on the following factors; the availability of NH_4^+ in the soil, and hence the deposition, the soil moisture and the temperature. Everything that is not taken up by the vegetation is apparently nitrified, generally NH_4^+ leaching only occur in regions with extremely high NH_4^+ deposition such as the Netherlands. Nitrification rate coefficient values can be derived from the data of Boxman et al., (1988), Johnsson et al., (1987), Jansson and Andersson, (1987), Nilsson and Grennfelt, (1988) and Sverdrup et al., (1990) and which reports that a low rate is $0.0018 \text{ kmol/m}^3 \text{ yr}$, a middle rate is in the range $0.0073\text{-}0.015 \text{ kmol/m}^3 \text{ yr}$ and a high rate is $0.015\text{-}0.03 \text{ kmol/m}^3 \text{ yr}$. This indicates that the nitrification rate coefficient has a value in the range $k_{nitr}=0.02\text{-}0.04 \text{ kmol/m}^3 \text{ yr}$, but the rate may be as low as $k_{nitr}=0.0015 \text{ kmol/m}^3 \text{ yr}$. The saturation coefficient has a value in the range $K_{sat}=4\text{-}6 \cdot 10^{-4} \text{ kmol/m}^3$. See also Tab. 7.5.2. The nitrification rates are sensitive to both temperature and soil moisture content, and must be adjusted for this prior to application. The

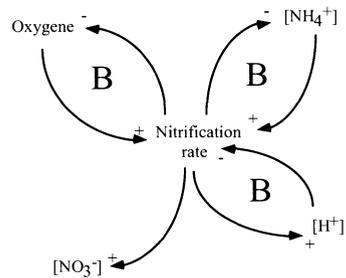


Figure 7.35: Causal loop diagram for the nitrification process. The nitrification rate depend on the ammonium concentration and is limited by substrate, access of oxygen and the retarding effect of acidity. All Swedish soils have the ability to start nitrification, and all ammonium that is not taken up by plants or denitrified will eventually be nitrified. This is proven by the fact that no Swedish soil is known to leach ammonium.

| Climate | Alpine | Boreal | Continental | Laboratory |
|--------------------------|---------------------------------------|--------|-------------|------------|
| Annual average °C | -1-5 | 5-8 | 8-12 | 25 |
| Nitrification rate | kmol m ⁻³ yr ⁻¹ | | | |
| Brown soils, pH 4.4-4.5 | 0.005 | 0.01 | 0.04 | 0.1 |
| Acid podsols, pH 4.5-4.8 | 0.0025 | 0.005 | 0.02 | 0.05 |

Table 7.22: *The approximate nitrification rate coefficient as estimated as averages from a number of determinations reported in the literature. After Sverdrup et al., (1990). The variation between soils appear to be mainly caused by climatic variations (temperature and soil wetness). The comparable rate used in the regional version of PROFILE was set at 0.07 kmol m⁻³yr⁻¹ at a reference temperature of 8°C. In PROFILE the rate is temperature-, water and pH adjusted for each site.*

given values assume a moisture saturation of 20%. The nitrification rate is influenced by the availability of moisture in the soil and increases with it. However nitrification is strictly aerobic and stops completely at full saturation of the soil with water. The function $f(\theta)$ adapted from the data of Russel et al., (1925) as shown in Figs. 7.37, may be given as:

$$f(\theta) = \begin{cases} 0 < \theta < 0.6 & ; 3.52 \cdot (\theta/\theta_m)^{2.25} \\ 0.6 < \theta < 1.0 & ; 0.0195 \cdot (\theta/\theta_m)^{-7.71} \end{cases} \quad (7.84)$$

The coefficients were derived from Fig. ???. This can also be seen as water absorption by the substrate and the desorbition of oxygen upon saturation of the soil with water, expressed as two Langmuir isotherms. The isotherms are different from the isotherms applicable to biomass degradation and tree uptake of nitrogen and base cations. This indicates that the mechanism of participation of water in nitrification is basically different from the mechanism by which water influences those other processes. Still it is tempting to aerobic nitrification does exist, but it was felt that it could be neglected on basis of information available in the literature at the present level of modelling ambition and scope. The nitrification rate is sensitive to the activity of the H⁺-ions in the soil solution, and an empirical relationship dependent on 0.25-th order of the H⁺-concentration in the soil solution, was suggested by Bossel et al. (1985), based on their review of the information available at that point in time in the literature. The expression is not based in any theoretical consideration, but rather on interpretation of empirical data. Bossel states that the data used is difficult to interpret, and that the relation is not firmly established. The retarding function describing the effect of toxic metals like Al and heavy metals is more complex and less well observed in laboratory or field experiments. However for trees the effect of Al could be shown to be connected to Al's ability to prevent the uptake of Mg and Ca, something that would also affect the uptake of these ions through the cell membrane of the nitrification bacteria in the soil. We assume the effects of Al and heavy metals to be separable and the Al effect is not mechanistically coupled to the heavy metal concentration, but that the heavy metal effect is interfered upon by the major constituents H⁺, BC²⁺ and Al³⁺ under acidic soil conditions. Here the concentration of Ca²⁺ and Mg²⁺ is called the concentration of BC²⁺. It has been shown that the Al retarding function can be derived from a postulated ion exchange mechanism, and the model can be confirmed by experimental data. The expression has the form:

$$f(BC/Al) = \frac{[BC^{2+}]^n}{[BC^{2+}]^n + K_{Al} \cdot [Al^{3+}]^m} \quad (7.85)$$

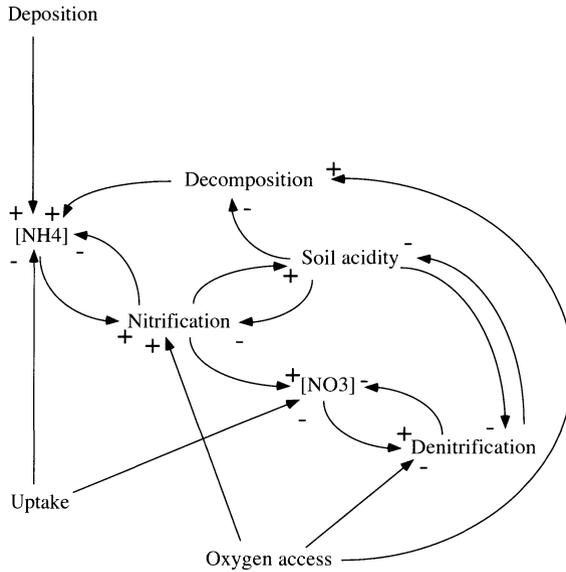


Figure 7.36: Causal loop diagram for the nitrogen transformation processes in the soil, including nitrification, uptake and denitrification. The causal loop diagram show the feedbacks included in the Sverdrup-Ineson model used for denitrification.

or if only a response to pH is available in the data:

$$f(BC/Al) = \frac{[BC^{2+}]^n}{[BC^{2+}]^n + K_H \cdot [H^+]^v} \tag{7.86}$$

where K_{Al} is related to the selectivity of base cations, aluminium and hydrogen ions to the microorganism surface. The exponents n and m depend on the reaction mechanism in the surface adsorption mechanism and the stoichiometry of that reaction (Sverdrup et al., 1992). For nitrification, the data indicate that the expression has the exponent $v=0.7$. The expression may imply that nitrification is fertilized by a calcium addition up to a certain level, and that the fertilization effect is stronger at lower pH values than at high pH values, which is also the qualitative observation from soil and forest liming projects in Sweden. The effect is not unlimited, and above a certain level, no more promoting effect can be gained. The heavy metals are also available in to low concentrations too low for them to act through the same surface competition mechanisms as Al. It is hypothesized that their action is metabolism interference through enzyme reaction poisoning inside the microorganisms. The poisoning effect on enzymes is proportional to the heavy metal concentration inside the bacteria, again dependent on the uptake of the heavy metals (Friberg et al., 1979) . However the adsorption sites for uptake to the bacteria are subject to competition from divalent base cations and the base cation competitions is negatively affected by hydrogen H-ions in acid soils. Accordingly, heavy metals cannot disrupt base cation uptake by competition, but base cations, Al or H-ions may disrupt or promote heavy metal uptake. On basis of this we can express the adsorption at the cell wall with the Gapon selectivity expressions This can be combined with the condition that heavy metal, Al, Ca and H are the only species adsorbed at these sites, to yield the retardation function for the effect of heavy metals and Al on nitrification rates. The data on the effect of Cu in the soil on nitrification has been shown by Tyler (1983). The full retardation function as based on an

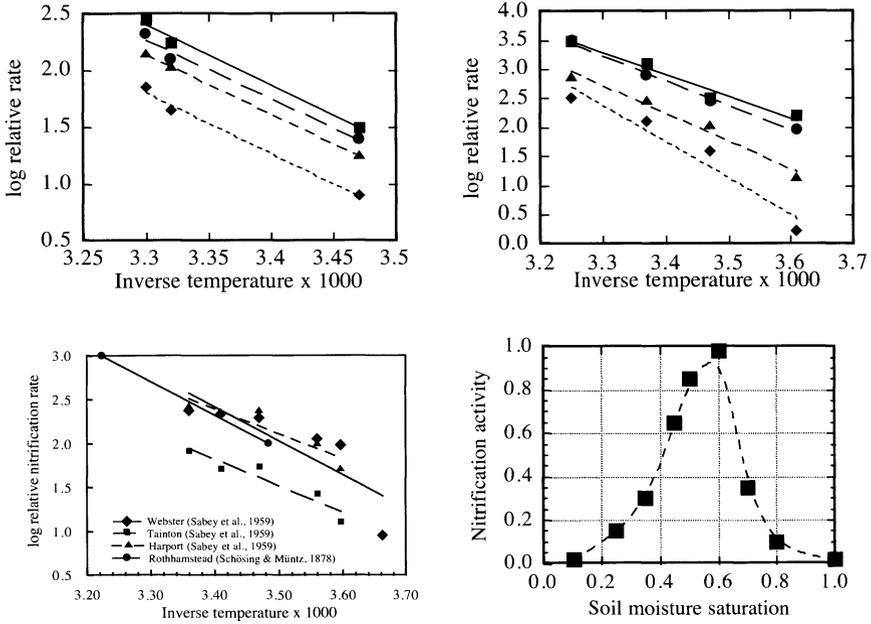


Figure 7.37: Data for the nitrification rate at different temperatures in American Lincoln and North Plate soil samples (Russel et al., 1925), and different British soil samples (Sabey et al., 1959). The last diagram, bottom, right, approximated dependency of nitrification in the soil as a function of the moisture saturation degree in the soil (Russel et al., 1925.)

equivalent exchange mechanism, can be simplified to:

$$f(M) = \prod_{i=1}^M \frac{1}{1 + k_{TOX_i} \cdot [M_i]^{m_i}} \tag{7.87}$$

The parameterization used data from Tyler (1974) and after rearrangement of the above equation, it is obtained:

$$\log(1/f(Cu) - 1) = \log k_{tox} + m \cdot \log [Cu] \tag{7.88}$$

was used to plot $\log(1/f(Cu) - 1)$ versus $\log [Cu]$ to get $\log k_{tox}$ as intercept and m as slope. We get $\log k_{tox} = -0.8 \pm 0.3$ and $m = 0.54 \pm 0.30$. The derived function has been put back into the data.

7.5.3 Nitrification dependence on temperature

The rate of nitrification, denitrification and uptake depends on the temperature, and data concerning this may be found in the literature. Russel, Jones and Baith (1925) investigated the rate of nitrification in two soils in jar tests. Their data were used to make a series of plots to determine the coefficients of the Arrhenius expression

$$r = r_0 \cdot 10^{A \cdot (1/(273+T) - 1/T_0)} \tag{7.89}$$

The slope is for the North American North Platte soil: 3770, 4253, 4803 and 6273, the average is $A=4774$. For the North American Lincoln soil we get the values: 5367, 5168, 5259 and 5369,

an average value of $A=5290$. For different British soils (Sabey et al., 1959) we get the values 3846, 3000, 2821 and 3831, with an average of $A=3374$. The observed value corresponds to an activation energy for the process of 64.6-101 kJ/mol. This is a typical value for processes which are chemical reaction rate controlled, indicating that the microbially mediated nitrification process may be rate limited by a respirational chemical reaction rate inside the bacteria. The data used is displayed in Fig. 7.37.

7.5.4 Discussion

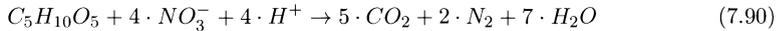
The presented model seem to explain the observed qualitative effects of heavy metals, Al and base cations (liming) on the nitrification rate. The temperature dependence of the overall rate could also be estimated. The nitrification rate is sensitive to temperature, and this quantification of the temperature dependence is necessary to have carried out. before any assessment of the effects in the climatic conditions can be made. A significant increase in temperature will increase the nitrification rate significantly. Nitrification will act on two sources available for NH_4^+ , degradation of biomass and deposition of NH_4^+ . The soils of the temperate zone of earth have very large stores of N in the reduced organic form (1,000-5,000 kg/ha), which would be available for nitrification if it were to be released by an increase in biomass decomposition. Such increases in biomass degradation is sometimes seen as temporal effects after clear-felling, which significantly change the hydrology and temperature regime of the upper soil for the first 2-5 years after a clear-felling. A rise in the ambient temperature of 5°C could potentially make this happened over all the forest surface, with devastating effects on the surface water quality and coastal marine environments. The Arrhenius factor for organic matter decomposition is approximately $A=5290$ (Sverdrup, de Vries and Henriksen, 1990), as compared to the Arrhenius factor for the nitrification reaction, $A=4480$ and for growth of trees, $A=3100$, base cation uptake, $A=4400$, and nitrogen uptake for coniferous forest, $A=3700$. Accordingly, if a global warming would occur, then biomass decomposition is going to produce more NH_4^+ than nitrification can consume, and the surplus would be more than what the vegetation could increase its uptake. Of course, tree uptake is both temperature dependent and nutrient soil solution concentration dependent, which would tend to increase the uptake enough to cope with reasonable increases. However this might be severely offset by the base cation long term availability due to chemical weathering in the forest soils of the temperate boreal zone. The soils on granitic bedrock the temperate zone have low weathering rates and could possibly not sustain a long term increase in forest growth. The rate increase of weathering with temperature, $A=3500-3800$, is equal to uptake temperature dependency, but less than nitrification and decomposition of biomass on the average (Sverdrup, 1990). These effects may potentially increase the NH_4^+ concentrations in the upper soil layers during the breakdown phase, leading to increased nitrification and a higher fraction of NH_4^+ to NO_3^- uptake by trees. The net effect would be increased soil acidification, which in due turn would slow the process of degradation and nitrification. The nitrification rate would not increase as much as the degradation rate, but this could possibly be compensated for by the increase in nitrification rate due to increased NH_4^+ soil solution concentration. Only further research could answer where the new soil steady state would be, probably with the help of sophisticated models in order to sort out the net result of all effects, counter-effects and couplings between processes. The ability of a soil to nitrify has often been related to soil type, C/N-ratio and climate. We think that this is in reality the dependence of the nitrification rate on soil pH, Al, base cation supply, temperature, and moisture conditions, as well as the availability of NH_4^+ in the soil. If the deposition is at or below the maximum uptake for the trees, all NH_4^+ and NO_3^- will be taken up, and the NH_4^+ concentration in the soil will always be very low. Since nitrification is dependent on concentration, the net rate will also be low. An indication of this is also that even very acid podsols tend to nitrify very well once they have been limed. At clear-felled sites where the temperature is increased, and the amount of NH_4^+ has been made much more available through increased biomass degradation, also tend to show nitrification comparable to that of a much more fertile soil.

7.6 Denitrification

The scope of this work is to determine an operational rate law for denitrification under field conditions in forest soils, taking into account large changes in climatic and chemical conditions, known to affect denitrification significantly. The rate law should be operable within the data available in national programs for mapping critical loads. It is an objective to (1) document the rate expression used in the PROFILE model, and (2) to document the expression used in the SMB and SMART model. The objective was to use accessible data compiled by Dr. Ineson at ITE, Merlewood, without making exhaustive and time-consuming literature and data searches.

7.6.1 Theory

Denitrification also occur by microbiologically mediated reactions. The net NO_3^- production in a system will accordingly be the difference between the total production by nitrification and the removal by denitrification. Denitrification is a multi-step reaction from NO_3^- over NO_2^- and N_2O to N_2 . There are several models available to model the reactions on a detailed level (McConnaughey and Bouldin, 1985a, b; McConnaughey et al., 1985, also giving kinetic rate coefficients and saturation concentrations. The overall reaction is:



It follows from the reaction stoichiometry that both nitrate and carbon is required in the substrate. This implies also that restricted decomposition of organic matter in the soil may lead to a restriction of denitrification. The role of sulfuric acidity, nitric acidity and acidity resulting from NH_4^+ reactions as the major anthropogenic constituents of acidic deposition is well established, as well as their acidifying effect on low weathering granitic soils. The role of nitric acidity, NH_4^+ and NO_3^- is somewhat more complicated than the more direct action of sulfuric acidity in soils. NH_4^+ is taken up by the trees in exchange for acidity, NO_3^- in exchange for alkalinity. NH_4^+ can be nitrified to NO_3^- , releasing two units of acidity for each molar unit of NH_4^+ nitrified. When N in the soil is denitrified, one unit of acidity is consumed for each equivalent unit N denitrified. The net acidity produced under the condition that no NH_4 is leached is:

$$\Delta \text{ANC}_N = N_u + N_{im} + N_{denit} - 2 \cdot N_{Nitr} \quad (7.91)$$

Thus kinetics of N uptake, nitrification and denitrification as well as the temperature dependence of these processes become important parts of soil acidification models. Uptake of NH_4^+ and nitrification produce one or respectively two units of acidity, nitrate uptake produce one unit of alkalinity, and net mineralization of organic matter to release NH_4^+ consume one unit of acidity. N fixation and NH_3 uptake is not influencing the alkalinity balance. In a simplified model, concerned only with the overall net rate, the denitrification rate has been described with the empirically modified Michaelis-Menten expression (Chen et al., 1983, Bossel et al., 1985, McConnaughey and Bouldin, 1985.; McConnaughey et al., 1985) as was also done for nitrification. Data supplied from unpublished sources, mostly from our colleagues, was used in estimating denitrification in forest soils. The data available at present is listed in Tab. 7.23. Denitrification has been shown to follow Michaelis-Menten kinetics in laboratory experiments, where the rate depend upon the aqueous solution concentration of nitrate:

$$r_{denit} = k \cdot \frac{[\text{NO}_3^-]}{K + [\text{NO}_3^-]} \quad (7.92)$$

The rate is modified by temperature, soil moisture and soil acidity:

$$k = k_0 \cdot \prod f_i \quad (7.93)$$

| Location | Total N deposition kg ha ⁻¹ yr ⁻¹ | Denitr. amount kg ha ⁻¹ yr ⁻¹ | Temp °C | pH | Moisture | Source |
|---|---|---|------------|-----|----------|--------------------------------|
| Spruce forest, Finland | 1 | 0.1 | 5 | 4.5 | 0.25 | Uomala et al., 1982 |
| Spruce forest, Northern Sweden | 2 | 0.1 | 2 | 4.2 | 0.25 | Ineson et al 1996 |
| Sitka plantation, Kershope, Britain | 7 | 3.2 | 10 | 4.5 | 0.25 | Ineson et al., 1991 |
| Beech forest soil, Michigan, USA | 10 | 1 | 22 | 4.3 | 0.25 | Groffman and Tiedje, 1989 |
| Silver fir forest, Massif Central, France | 14 | 4.4 | 11 | 4.3 | 0.25 | Lensi et al., 1991 |
| Ash forest, Klosterhede, Denmark | 20 | 3 | 13 | 6.1 | 0.1-0.5 | Struwe and Kjöllér, 1991 |
| Alder forest, Klosterhede, Denmark | 20 | 4.9 | 13 | 7.5 | 0.1-0.5 | Struwe and Kjöllér, 1991 |
| Spruce forest, Haldon, Britain | 21 | 5 | 7 | 4.5 | 0.25 | Willison and Anderson, 1991 |
| Spruce forest, Haldon, Britain | 21 | 8 | 7 | 5.3 | 0.25 | Willison and Anderson, 1991 |
| Mire, Southern Sweden | 23 | 3 | 6 | 4.5 | 0.9 | Ineson et al 1996 |
| Spruce forest, Southern Sweden | 23 | 0.5 | 6 | 3.9 | 0.25 | Ineson et al 1996 |
| Spruce forest, Southern Sweden | 23 | 0.1 | 6 | 3.8 | 0.25 | Ineson et al 1996 |
| Spruce forest, Southern Sweden | 23 | 0.2 | 6 | 3.9 | 0.25 | Ineson et al 1996 |
| Spruce forest, Southern Sweden | 23 | 1 | 6 | 4.1 | 0.25 | Ineson et al 1996 |
| Beech forest, Southern Sweden | 25 | 6 | 6 | 4.8 | 0.25 | Ineson et al 1996 |
| Beech forest, Southern Sweden | 29 | 7 | 6 | 4.9 | 0.25 | Ineson et al 1996 |
| Spruce forest, Göttingen, Germany | 35 | 7 | 9 | 4.1 | 0.25 | Ulrich, 1992 |
| Beech forest, Göttingen, Germany | 35 | 8 | 9 | 4.1 | 0.25 | Beese et al., 1991 |
| Spruce forest, Innsbruck, Austria | 45 | 12 | 6 | 4.8 | 0.25 | Henrich and Haselwandter, 1991 |
| Oak forest, Tenkink, Netherlands | 80 | 20 | 9 | 4.2 | 0.25 | Tietema et al., 1991 |
| Spruce forest, Innsbruck, Austria | 90 | 15 | 6 | 4.8 | 0.25 | Henrich and Haselwandter, 1991 |

Table 7.23: *In situ* measurements of denitrification rates. The difference in values reflect differences in soil conditions. Data were supplied unpublished from colleagues at the Institute of Terrestrial Ecology at Merlewood, Great Britain, Finnish Agricultural University at Åbo, University of Forest and Agricultural Sciences at Umeå, Laboratory for Environmental Sciences at the Danish Technical University, University of Göttingen, University of Wageningen, Industrial Forest Research Institute at Uppsala and University of Lund, and may be subject to substantial later revisions. Soil moisture saturation was not always available and is an approximate estimate for most sites.

f_i is a number of multiplicative modifying functions taking into account the effect of temperature, chemical conditions and soil moisture conditions. Chemical conditions in terms of soil acidity as expressed by pH and Al concentrations may influence the rate. Assuming no ammonium ever to leach from the soil, we may write:

$$[NO_3^-] = \frac{N_l}{Q} \quad (7.94)$$

The kinetic expression may be rearranged by inserting Eq. 7.94 in Eq. 7.92:

$$r_{denit} = k \cdot \frac{N_l}{K \cdot Q + N_l} \quad (7.95)$$

The net leaching available for denitrification may be approximated from a mass balance:

$$N_l = N_{dep} - N_u - N_{im} \quad (7.96)$$

The kinetic expression may be rewritten to the Sverdrup-Ineson expression:

$$r_{denit} = k \cdot \frac{N_{dep} - N_u - N_{im}}{K \cdot Q + N_{dep} - N_u - N_{im}} \quad (7.97)$$

This is an approximative expression, based on fluxes instead of proper concentrations.

7.6.2 Data

In situ measurements of denitrification rates have been listed in Tab. 7.23. The difference in values reflect differences in soil conditions between the sites. Data were supplied unpublished from

colleagues at the Institute of Terrestrial Ecology at Merlewood, Great Britain, Finnish Agricultural University at Åbo, University of Forest and Agricultural Sciences at Umeå, Laboratory for Environmental Sciences at the Danish Technical University, University of Göttingen, University of Wageningen, Industrial Forest Research Institute at Uppsala and University of Lund, and may be subject to substantial later revisions. Soil moisture saturation was not always available and is an approximate estimate for most sites.

7.6.3 Results

Data from Struwe and Kjöllér (1991) was used to determine the coefficients of the Michaelis-Menten expression. They determined the in situ nitrification rate in an alder (*Alnus glutinosa*) stand and an ash (*Fraxinus excelsior*) stand. Both soils are waterlogged most of the year, the determinations occurred at temperatures of 14°C in August, 10°C in October and 2°C in January. In situ additions of nitrate was applied to the soil in order to investigate the reaction of the rate to added nitrate. The ash site had a of pH 7.5, the alder site pH 6.1 in the soil solution.

The observed in situ field rate was plotted versus the soil solution nitrate concentration, as is shown in Fig. 7.42. The plot can be used to determine the coefficients of the Michaelis-Menten expression. Additional experiments were carried out in slurries, the results are shown in Fig. 7.42. The release rate of $1 \cdot 10^{-9} \text{ g N}_2 \text{ g}^{-1} \text{ day}^{-1}$ was translated to be equivalent to a denitrification of $1 \text{ kg N ha}^{-1} \text{ yr}^{-1}$. This is a conservative estimate. The rate coefficient is normalized to pH 5 using Eq. 7.98, and to soil moisture saturation of 0.2 using Eq. 5. The soil moisture saturation of a "mostly waterlogged" soil is difficult to estimate, but was set at 0.1 in August, 0.3 in October and 0.5 in January in this calculation. In the results emanating from the experiments, the Michaelis-Menten saturation constant remains the same regardless of experiments. This constant is however not affected by soil moisture saturation or pH, and depend on concentrations alone. The rate coefficient vary somewhat between in situ observation and slurry experiments. This comes partly from the difficulty in assessing a soil moisture saturation value to the in situ experiments, and the difficulty in assessing a pH value to the slurry experiments.

Several factors modify the rate coefficient. The most important identified so far are temperature, soil moisture and soil solution pH. The modifier function is assumed to consist of three separable and independent response functions:

$$\prod f_i = g(T) \cdot y(w) \cdot f(pH) \quad (7.98)$$

Each of these response functions will be quantified and parameterized below. The information on soil moisture saturation for the sites listed in Tab. 7.23 was mainly of qualitative nature, and the values given are approximated values. This is one the more uncertain parts of the data used at present. Soil moisture saturation was generally given as "normal forest soil" or "wet soil" or "moist" or "waterlogged". The soil moisture saturation for normal forest soils was set to 0.2, moist and wet it was set to 0.3 and 0.4, for waterlogged soils it was set to 0.5. In slurry

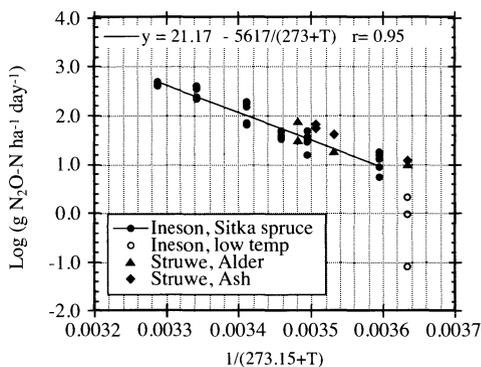


Figure 7.38: Denitrification dependence on temperature as determined by Ineson et al., (1991) and Struwe and Kjöllér, (1991).

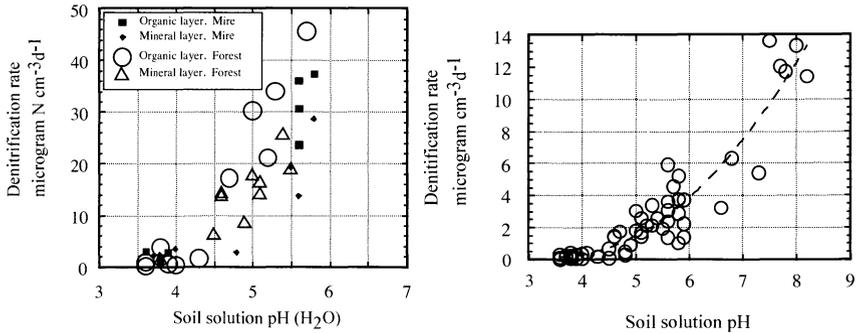


Figure 7.39: Different studies showing pH dependence of the denitrification rate dependence (Müller et al., 1980).

experiments it was set to 1.0. Thus it should be realized that the soil moisture information used contain a considerable amount of uncertainty. The soil moisture relation was suggested by Bossel et al., (1985) to have a form similar to an adsorption isotherm:

$$x(w) = \frac{k_w \cdot w}{1 + k_w \cdot w} \quad (7.99)$$

x is the fraction of added nitrogen denitrified in experiments. The soil moisture saturation rate modifying function has the distinct appearance of a Langmuir absorption isotherm, indicating that the moisture relationship is caused by some physical water adsorption process in the soil. A similar expression is valid for the effect of moisture on organic matter decomposition (Jönsson et al., 1994). w is the soil moisture saturation, $k_w=0.96$. Making the function assume the value 1.0 at a soil moisture saturation of 0.2, yield the expression after division with the functional value at 0.2:

$$y(w) = \frac{x(w)}{x(0.2)} = \frac{5.96 \cdot w}{1 + 0.96 \cdot w} \quad (7.100)$$

$y(w)$ is the moisture modifier function. Denitrification increase significantly as soon as the soil is waterlogged, and a substantial amount of NO_3^- may denitrify in discharge areas in catchments, and thereby decrease the amount N leached to surface waters. This would explain some of the discrepancies sometimes seen in forests stands between deposition, uptake and leached amounts. The soil moisture dependency was taken from Nömmik (1956) and Bremner and Shaw, (1958). The rate follow the Arrhenius relation between 0°C and 35°C . Data was found in the literature (Ineson et al., 1991; Struwe and Kjöllér, 1991; Bremner and Shaw, 1958) from which the dependency could be determined as shown in Fig. 7.30. The Arrhenius factor was determined from the slope of the line; $A=5620$, the expression is:

$$g(T) = 10^{A/281 - A/(273+T)} \quad (7.101)$$

The expression is valid in the temperature range from -2°C to $+60^\circ\text{C}$, according to the cited literature.

Denitrification in soils is sensitive to soil acidity. Two literature sources were used to derive data on which the following empirical function was based (Bremner and Shaw, 1958; Mueller et al., 1980), it represents observed denitrification rate in $\mu\text{g N cm}^{-1}\text{day}^{-1}$ as a function of soil solution pH:

$$r_{denit,pH} = 6 \cdot pH^2 - 42.2 \cdot pH + 75.7 \quad (7.102)$$

| Experiment type | Observed rate coefficient kg N ha ⁻¹ yr ⁻¹ | Temperature °C | Soil moisture saturation | pH | Michaelis-Menten rate coefficient kg N ha ⁻¹ yr ⁻¹ | Mikaelis-Menten saturation constant kmol m ⁻³ |
|---------------------------|---|----------------|--------------------------|-----|---|---|
| Alder 1, in situ, August | 170 | 14 | 0.2 | 6.1 | 22 | 2*10 ⁻⁴ |
| Alder 2, in situ, October | 40 | 10 | 0.3 | 6.1 | 74 | 2*10 ⁻⁴ |
| Alder 3, in situ, January | 40 | 2 | 0.5 | 5.6 | 30 | 2*10 ⁻⁴ |
| Alder 1, slurry, August | 5500 | 14 | 1.0 | 6.1 | 248 | 2*10 ⁻⁴ |
| Alder 2, slurry, October | 11500 | 10 | 1.0 | 6.1 | 97 | 1*10 ⁻⁴ |
| Alder 3, slurry, January | 5500 | 2 | 1.0 | 5.6 | 2640 | 1*10 ⁻⁴ |
| Ash 1, in situ, August | 400 | 14 | 0.2 | 7.5 | 23 | 2*10 ⁻⁴ |
| Ash 2, in situ, October | 140 | 10 | 0.3 | 7.5 | 11 | 2*10 ⁻⁴ |
| Ash 3, in situ, January | 40 | 2 | 0.5 | 7 | 10 | 2*10 ⁻⁴ |
| Ash 1, slurry, August | 2900 | 14 | 1.0 | 7.5 | 60 | 2*10 ⁻⁴ |
| Ash 2, slurry, October | 7500 | 10 | 1.0 | 7.5 | 270 | 1*10 ⁻⁴ |
| Ash 3, slurry, January | 7500 | 2 | 1.0 | 7 | 1330 | 5*10 ⁻⁴ |

Table 7.24: *The denitrification rate coefficient and saturation constant in the Michaelis-Menten expression, as calculated from estimated soil conditions and observed denitrification rates in field and laboratory experiments.*

The expression can be normalized to become 1.0 at soil solution pH 5, this is obtained by dividing by 14.7, the functional value at pH 5:

$$f(pH) = \frac{r_{denit,pH}}{r_{denit,pH5}} = 0.41 \cdot pH^2 - 2.87 \cdot pH + 5.15 \quad (7.103)$$

Both studies were consistent concerning the dependency of nitrification on pH. The data from Mueller et al., (1980) on denitrification in different types of soil is shown in Fig. 7.23. The data indicate that there is no significant difference between the dependence on pH in different soil layers. Nor is there any large difference between a forest soil and a mire. This is surprising, as one would expect denitrification in a waterlogged mire to be significantly higher than in a forest soil. All data available from Müller et al., (1980), Willison and Anderson, (1990) and Bremner and Shaw, (1958) have been put together in Fig. 7.40.

Most plants and soil microorganisms are sensitive not only to H⁺-ions but also to Al. The molecular mechanism for inhibition in microorganisms would be competitive ion exchange at outer membranes engaged in nutrient uptake, if we were allowed to make the analogy with plants and aquatic animals. Then the rate modifier should have a form such as:

$$f(pH) = \frac{r_{denit}}{r_0} = \frac{[BC^{2+}]^n}{[BC^{2+}]^n + K_H \cdot [H^+]^m} \quad (7.104)$$

If the concentrations of base cations like Ca and Mg, are unknown, the expression would degenerate to:

$$f(pH) = \frac{1}{1 + K_H \cdot [H^+]^m} \quad (7.105)$$

By plotting log(1/Rate-1) versus pH, as has been shown in Fig. 7.39, we can determine the rate coefficient $r_0=150 \mu\text{g N cm}^{-1}\text{day}^{-1}$, and more important, the inhibition coefficient $\log(K_H)=4.7$ and $m=0.7$. This is close enough to 1.0 for us to suggest that the "unspecific" response type is

involved (Sverdrup and Warfvinge, 1993; Jönsson et al., 1994). The inhibition coefficient include the base cation and Al concentrations since they were unknown in the experiment. The straight line obtained in Fig. 7.39 and the good correlation to a straight line is a strong indicator for H^+ surface adsorption to be the mechanism of soil acidity inhibition. Such an inhibitor mechanism could be ascertained for soil organic matter decomposer microorganisms (Jönsson et al., 1994). There is a possibility that increasing the concentration of base cations like Ca^{2+} , K^+ and Mg^{2+} , will promote denitrification as well as other microbial processes in the soil, and possibly counter-react any toxic effect of Al, as is the case with decomposition and tree root uptake of nutrients. Further research is needed to test this hypothesis.

Empirical data as listed in Tab. 7.23, indicate that approximately 25% of the deposition of N gets denitrified. Quantitative values for denitrification in normal forest soils are elusive, denitrification rates in soils is often assumed to be negligible as anaerobic conditions are generally required. The values have all been adjusted to 8°C, to remove temperature effects. The rate was also corrected for soil moisture and soil pH. There was no data available on soil solution concentrations of nitrogen, and total nitrogen deposition was used as a substitute variable. This implies that we can estimate the denitrification base rate rate from observed data, removing temperature, soil moisture and soil chemistry effects. The base rate was calculated as the denitrification rate at 8°C, soil moisture saturation of 0.2 and soil pH 5.0. The modifying functions were renormalized in such way that they assume the value 1.0 at the conditions described above. These values were chosen because they are typical values for soil temperature and soil moisture in Northern Europe, and because soil pH 5.0 will be a typical solution pH for forest soils not receiving any acid deposition, e.g. when critical loads for acidity are met. The base rate was calculated from:

$$r_{base} = \frac{r_{obs}}{\prod f_i} \quad (7.106)$$

r_{base} is the base rate. The modifying functions used to normalize the denitrification rate to the conditions described above, are:

$$\prod f_i = (10^{20-5620/(273+T)}) \cdot \left(\frac{5.96 \cdot w}{1 + 0.96 \cdot w} \right) \cdot (0.41 \cdot pH^2 - 2.87 \cdot pH + 5.15) \quad (7.107)$$

The field rate can be described with an empirical rate law, modified to accommodate that there appear a minimum rate of approximately 0.1 kg N ha⁻¹yr⁻¹ under field conditions:

$$r_{denit} = \max \left(r_{min}, \left(k \cdot \frac{N_l}{K + N_l} \right) \right) \quad (7.108)$$

This expression is used for critical load calculations, since N_l is determined by the critical limit for leaching. For calculation of present rate, a slightly different expression can be used. The rate for any soil with any soil pH, temperature or soil moisture saturation, can be estimated from the rate expression, applying the rate modifiers:

$$r_{denit} = \max \left(r_{min}, \left(k_0 \cdot \frac{(N_{dep} - N_u - N_{im})}{K + (N_{dep} - N_u - N_{im})} \right) \right) \cdot \prod f_i \quad (7.109)$$

where N_{dep} is total nitrogen deposition, N_u is tree nitrogen uptake.

$r_{min}=0.7$ kg N ha⁻¹yr⁻¹ is a minimum denitrification rate under optimal conditions (0.05 keq ha⁻¹yr⁻¹), the first order rate coefficient value is $k_0=25$ kg N ha⁻¹yr⁻¹ (1.8 keq ha⁻¹yr⁻¹) and the Michaelis-Menten half rate saturation coefficient $K=40$ kg N ha⁻¹yr⁻¹ (2.86 keq ha⁻¹yr⁻¹). On the average immobilization in the region covered by our database is $N_{(im)}=8$ kg N ha⁻¹yr⁻¹. This explains why relatively few sites at present leach nitrogen. k_0 is the rate coefficient k divided by the functional value of pH 5 in the non-normalized pH modifier and the functional value of the moisture modifier at $w=0.2$. Other kinetic expressions are also possible within the data set, but the accuracy of the compiled data was not considered to support differentiation arising from

| Experiment type | Slurry/in situ rate | Ratio |
|-----------------|---------------------|-------|
| Alder, August | 248/22 | 11.3 |
| Alder, October | 97/74 | 1.3 |
| Alder, January | 2640/30 | 88 |
| Ash, August | 60/23 | 2.6 |
| Ash, October | 270/11 | 24 |
| Ash, January | 1330/10 | 133 |
| Average | | 43 |

Table 7.25: The ratio between in situ and slurry denitrification rate coefficients after correction for temperature, soil moisture and soil pH.

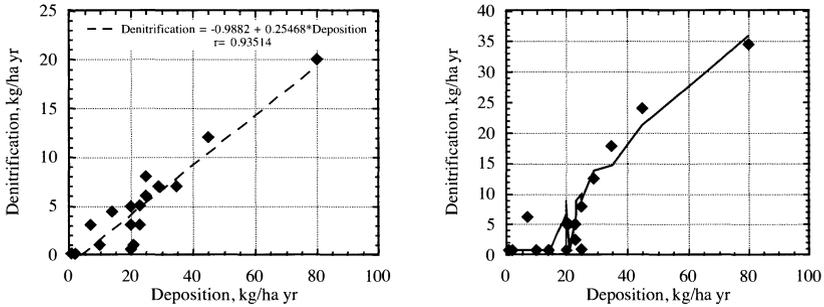


Figure 7.40: The dependence of denitrification on nitrogen deposition as observed without any modifications (top), and after adjustment for differences in temperature, soil moisture and soil pH between sites (bottom). The value at a deposition of 90 kg/ha yr is very uncertain. The drawn line is the predictions for field denitrification rate by the model. Data was compiled from unpublished values from research colleagues and from the grey literature.

such small differences. The back-calculation using the empirical formula, yield a correlation of $r^2=0.87$ when compared to the original measurements adjusted for temperature, soil moisture and soil acidity. This is the same correlation as for plotting denitrification data adjusted for temperature, soil moisture and soil acidity directly against deposition ($r^2=0.87$).

7.6.4 Discussion

Several problems remain in this study. The denitrification activity decrease down through the soil profile. This may be caused by several alternative mechanisms. The increase of dissolved aluminium, highly toxic to plants and microorganisms, may be preventing denitrification. The decrease of nitrogen concentration down through the soil profile may be part of the explanation, but data seem to suggest additional effects. The decrease of organic matter in deeper layers may be a cause of carbon limitation for the denitrification process. In the experiments, nitrate was added to soils as KNO_3 . The added K-ions will offset adsorbed H^+ -ions and Al^{3+} -ions and decrease soil solution pH significantly. Thus solution pH was not

constant in the slurry experiments. The effect on pH by temperature could only be approximated, this is reflected in the changing pH values in Tab. 7.24 The discrepancy between in situ measurement and slurry experiments is a well known phenomenon according to Struwe and Kjöllér (1991). We think the explanation may be explained by a well known mechanism. In the soil, stagnant conditions can be said to prevail. It can be shown that the effective distance of diffusion will be approximately equal to the average soil particle diameter (Sverdrup and Bjerle, 1982; Warfvinge, 1988). In podzolic forest soils, the soil structure will be dominated by particles in the range from 10-200 μm . A typical example would be 5-10% clay (0-2 μm), 20-30% silt (2-63 μm) and the rest sand (63-2000 μm). This would imply diffusion distances of 50-200 μm . A part of the volume will be occupied by non-permeable solids, and this reduces the diffusivity approximately to 1/3 (Sverdrup et al, 1983; Warfvinge 1988). In slurry experiments the mixture of soil and solution is well stirred (and sometimes shaken..). This greatly enhances mass transfer conditions and mass transport will only be limited by diffusion through a boundary layer around each particle of approximately 10 μm thickness.

Thus the rate of denitrification in slurry experiments should be expected to be 15-60 times larger than in situ rates. The range of ratios observed in Tab. 7.25 fall in the range 1.3-133. The approach taken here ignores the different types of denitrifying organisms in the soil and that the denitrification process may be incomplete with respect to nitrogen gas. This is known to be affected by the denitrifying conditions. The model was tested on the deposition and recorded pH, temperature and soil moisture for the different sites. The overall accuracy in the model calculations appear to be in the order of $\pm 45\%$. The outlier is based on laboratory results from Heinrich and Haelwandter, (1991) and can be ignored as an uncertain, unduplicated laboratory experiment, which not necessarily can be extrapolated to field conditions. The rate law derived here is empirical and based on data from field observations. Strictly seen, a mechanistic model should have been based on soil concentrations, since that is what the microorganism can register and respond to. However, such data were simply not available within the regional databases available for calculation of critical loads, and other ways had to be tried. Deposition will be proportional to soil solution concentrations, especially under high deposition conditions, possibly one reason why the empirical relation seem to work. The field experiments yield a rate coefficient of 28 kg N ha⁻¹yr⁻¹ and a half rate saturation coefficient of $K=2 \cdot 10^{-4}$ kmol m⁻³. The slurry experimental rate, performed at water saturation yield an average rate coefficient of $k=774$ kg N ha⁻¹yr⁻¹ and a saturation coefficient of $K=2 \cdot 10^{-4}$ kmol m⁻³. Converting the slurry rate to field rate using the ratio 43 between slurry and in situ experiments, yield $k=18$ kg N ha⁻¹yr⁻¹. Assuming the difference to arise from mixing conditions, allow us to estimate from soil liming diffusion distances of 0.2 mm, indicating that the ratio should be 20-30, in turn yielding a rate coefficient of $k=26-39$ kg N ha⁻¹yr⁻¹. The amount subtracted from the deposition in the kinetic expression, may be a result of soil N concentrations not being directly 1:1 proportional to deposition. The soil flux

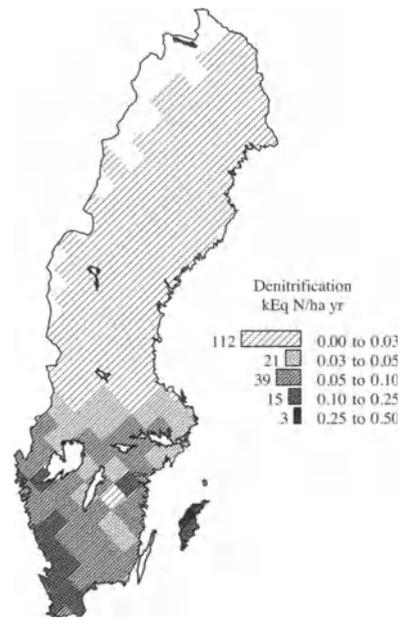


Figure 7.41: Calculated denitrification rate in keq N/ha yr for Swedish forest soils using this model in the PROFILE model. Input data was taken from the Swedish Forest Inventory. The data applies to the situation in 1991-1993.

| Conditions | Rate coefficient | Michaelis Menten halfrate coefficient | Empirical half-rate coefficient |
|----------------------------|--|---------------------------------------|--|
| Units | k_0 kgN ha ⁻¹ yr ⁻¹ | K kgN m ⁻³ | K-Q kgN ha ⁻¹ yr ⁻¹ |
| Regional field observation | 25 | $5.7 \cdot 10^{-3}$ | 40 |
| Laboratory experiments | 18 | $2.8 \cdot 10^{-3}$ | 20 |
| In situ observation | 28 | $2.8 \cdot 10^{-3}$ | 20 |
| Average value | 24 | $3.8 \cdot 10^{-3}$ | 27 |

Table 7.26: *Denitrification rate coefficients and Michaelis Menten half rate saturation coefficients.*

of nitrogen available for denitrification will be offset by an amount equivalent to what has been consumed by other processes such as uptake. Total uptake in a spruce forest stand can amount to 10-20 kg N ha⁻¹yr⁻¹, but generally 3-6 kg N ha⁻¹yr⁻¹ will be returned by litterfall and 3-6 kg N ha⁻¹yr⁻¹ will be returned by in debris left after cutting the trees. 1-10 kg N ha⁻¹yr⁻¹ will be returned as litter, but not decomposed, and thus become immobilized in the soil, unavailable for soil microorganisms. The estimation of the immobilized amount is rather uncertain. r_{min} corresponds to a minimum denitrification rate that would always occur whenever nitrogen is transformed by microorganisms in the soil.

The amount subtracted from the deposition in the rate expression; under present (1990) conditions uptake and immobilization is close to 20 kg N ha⁻¹yr⁻¹. This is close to the deposition level at which nitrogen leaching from soils is observed to start to occur more frequently (Fig. 7.39). This further reinforces the idea that substantial denitrification requires a substantial concentration of nitrogen in the soil solution to occur. The strong dependence of the rate on soil pH may seem puzzling, but the data is rather consistent in laboratory experiments. It is also consistent with similar studies of decomposition of organic matter (Jönsson et al., 1994). This implies that there is a large denitrification potential hidden in acidified forest soils today, which could transfer larger amounts of nitrogen if the pH were to increase due to soil liming or substantial reductions in acidic deposition. Many forest soils are acid today due to acidic deposition, and would increase in pH more than one unit if the anthropogenic acid deposition was reduced to pre-industrial levels, this applies to many podsoils and brown soils in Scandinavia (Sverdrup et al., 1990; Warfvinge et al., 1993; Sverdrup and Warfvinge, 1993; Warfvinge et al., 1993). Under critical loads, the deposition will in general be smaller, and the amount of excess N may be small. The the expression may degenerate to a first order expression:

$$r_{denit} = (r_{min} + k_0 \cdot N_t) \cdot \prod f_i \quad (7.110)$$

It appears that it does not matter much whether the denitrification rate is correlated to deposition or calculated leaching. Both yield the same goodness of fit of the empirical expressions as long as the effect of temperature, soil moisture and soil acidity is accounted for. Thus the following appear as good as the one mentioned above:

$$r_{denit} = k_d \cdot N_{dep} \cdot \prod f_i \quad (7.111)$$

where $k_d=0.25$. The kinetic expression based on the nitrate concentration and parameterized using the Danish experiments, was inserted in the PROFILE model (Sverdrup and Warfvinge,

1993; Warfvinge et al., 1993), and the denitrification rate calculated for Swedish forest soils. The calculated rates have been shown in Fig. 7.41, units are in $\text{keq N ha}^{-1}\text{yr}^{-1}$.

7.6.5 Conclusion

The dependence of denitrification on soil pH, also implies that the critical load of nitrogen to forest soils is non-linearly connected to sulphur deposition. A higher sulphur deposition will lower the critical load for nitrogen, a lower sulphur deposition could potentially increase the critical load for nitrogen significantly, especially for wet and waterlogged soil types. This is a possibility for getting additional benefits from a substantial sulphur deposition reduction, but poses a computational problem for calculation of critical loads, due to the non-linearity. The evaluation of the data suggests that N immobilization in these soils is $8 \text{ kg N ha}^{-1}\text{yr}^{-1}$ on the average. This is a significant part of the deposition, and stresses that immobilization is important for the critical loads of nitrogen. The use of nitrogen deposition in the kinetic expression is a large simplification, introducing significant uncertainties. A calculation based on soil solution concentrations would be preferable when possible. Leaching of nitrogen from soils increase when the deposition is larger than 10-15 kg N/ha yr , close to the deposition at which also the denitrification rate increases significantly. This can be explained easily if the denitrification rate depend on the soil solution concentration of nitrate. The dependence of denitrification should probably incorporate different promoting cations like Ca and Mg and general inhibitors such as Al in the soil solution, in addition to soil pH. More research is required to produce the necessary data. The non-linearity implies that multi-layer, integrated models should be used for calculation of critical loads whenever denitrification is important. This to ensure that the necessary feedback loops can be integrated into the calculation. The simplified calculation method suggested so far (Gundersen et al., 1993) will probably suffice for drier or more acid forest soils. The use of the presently suggested expressions do give regional denitrification rates for Sweden appearing to be reasonable for use in critical loads calculations.

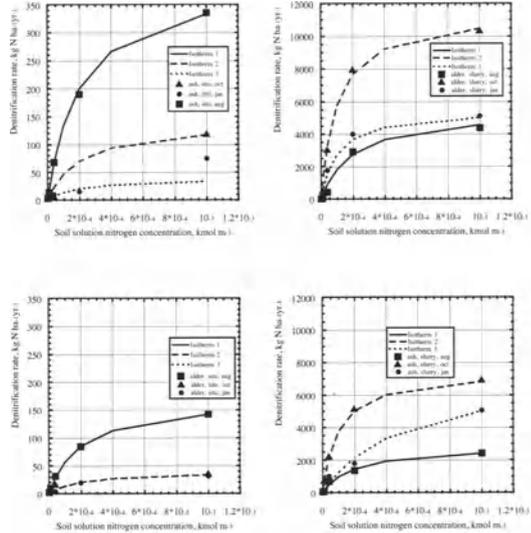


Figure 7.42: (1) *In situ* observed field denitrification rate as a function of soil solution nitrate concentration for soils from the alder forest site. (2) Observed denitrification rates in slurry experiments as a function of soil solution nitrate concentration for soils from the alder forest site. (3) *In situ* observed field denitrification rate as a function of soil solution nitrate concentration for soils from the ash forest site. (4) Observed denitrification rates in slurry experiments as a function of soil solution nitrate concentration for soils from the ash forest site.

7.7 The SAFE model

7.7.1 Mass balance equations

The change in soil solution chemistry and the subsequent change in the distribution of elements on the exchange matrix are calculated by means of conservation equations, i.e. mass balance equations. A mass balance is based on the fact that mass and energy are indestructible entities. A generic mass balance of substance Y in molar units over a continuously stirred tank reactor (CSTR), the mixing model used for each soil horizon in SAFE and PROFILE, may be written as inputs must balance outputs and internal accumulation:

$$Q_0[Y]_0 + r_Y V = Q[Y] + \frac{d(V[Y])}{dt} \quad (7.112)$$

where Q is the volumetric flow in $m^3 s^{-1}$, $[Y]$ the concentration of Y in $kmol m^{-3}$, r_Y the production term, i.e. the overall rate at which Y is produced within the soil horizon, in $kmol m^{-3} s^{-1}$, V the water volume in m^3 and where index 0 denote the conditions in the water entering the soil layer and no index denote the conditions within the layer. Note that in a CSTR the conditions within the systems are identical to the conditions in the outflow from the system. The SAFE and PROFILE models are one dimensional models and the generic mass balance is therefore rewritten as

$$\frac{d[Y]}{dt} = \frac{1}{z\theta} \left(Q_0[Y]_0 - (Q + z \frac{d\theta}{dt})[Y] + r_Y \right) \quad (7.113)$$

where Q is the flow in $m^3 m_s^{-2} s^{-1}$, z the layer thickness in m_s , θ the volumetric water content in $m^3 m_s^{-3}$ and r_Y the production term in $kmol m^{-3} s^{-1}$. If volumetric water content (θ) varies, changes in solution chemistry will occur because of dilution/concentration. This is quantified in the equations by the $z \frac{d\theta}{dt}$ term. In the current version of SAFE, changes in θ are neglected since no hydrological model currently is included. The hydrogen ion is treated as dependent on the variable acid neutralizing capacity (ANC). In the mass balance for ANC, the unit for concentration is in $kmol_c m^{-3}$ and the production term is in $kmol_c m^{-2} s^{-1}$.

The base cations calcium, magnesium and potassium are grouped together on an equivalent basis and are regarded to react as a divalent component, Bc. A base cation is a cation to a strong base, e.g. Sodium is the cation to the strong base sodium hydroxide (NaOH). In literature regarding weathering rates, the term base cations usually includes Ca, Mg, K, Na. As Na is the only base cation not considered to be a major nutrient for plant uptake, Na is sometimes excluded when base cations are defined in literature concerning plant uptake and ecosystem health. To minimize the confusion, base cations excluding Na is abbreviated Bc, while base cations including Na is abbreviated BC. In SAFE, Na is neglected as an exchangeable species as well as a nutrient, and is assumed to behave as a conservative element. The production terms in equation 7.113 are given by:

$$r_{ANC} = 2W_{Bc} + W_{Na} + 2r_{E_{Bc}} - 2U_{Bc} + U_N + 2NM_{Bc} - NM_N \quad (7.114)$$

$$r_{Bc} = W_{Bc} + r_{E_{Bc}} - U_{Bc} + NM_{Bc} \quad (7.115)$$

$$r_{NO_3^-} = U_N - NM_N \quad (7.116)$$

$$r_{Cl^-} = 0 \quad (7.117)$$

$$r_{Na^+} = W_{Na^+} \quad (7.118)$$

$$r_{SO_4^{2-}} = NM_S \quad (7.119)$$

where $r_{E_{Bc}}$ is the contribution from cation exchange (equation 7.149 and 7.150), W is weathering, U is uptake and NM is net mineralization of the respective species, all units in $\text{kmol}_c \text{m}^{-2} \text{s}^{-1}$. In addition to the mass balances for soil solution, one mass balance is needed for Bc on the cation exchange surface

$$\frac{dE_{Bc}}{dt} = -\frac{2r_{E_{Bc}}}{z\rho\text{CEC}} \quad (7.120)$$

where CEC is the cation exchange capacity in $\text{kmol}_c \text{kg}^{-1}$, E_{Bc} the equivalent base saturation fraction and ρ is the field moist bulk density in kg m^{-3} . An increase in base saturation thus corresponds to a withdrawal of ANC (equation 7.114) and Bc (equation 7.115) from the soil solution.

Biological feedbacks

The vegetation of the forest stand is in most cases very important for the cycling of nutrients, such as Bc , phosphorus and N . In this way it affects the soils balance of acidity what itself is a strong driving force in many soil processes. Major biological processes in this context are the growth of plants, decomposition of organic matter, nitrification and denitrification. The current version of the *SAFE* and *PROFILE* models does not model vegetation growth, nutrient cycling, decomposition or immobilisation explicitly. Instead it is up to the model user or another model, e.g. the *MAKEDEP* model, to provide time-series on net nutrient uptake, canopy exchange, litter fall and net mineralization. Net mineralization is the net result of immobilisation and decomposition and can thus be either positive or negative. Canopy exchange is defined as positive when nutrients are exuded from the canopy and negative when nutrients are taken up by the canopy. Net mineralization is assumed to occur in the top layer only. What actually is being taken up by vegetation in the rooting zone, the gross uptake, is calculated as:

$$\text{GU} = \text{NU} + \text{CE} + \text{LF} \quad (7.121)$$

where GU is gross uptake, NU is net uptake, CE is canopy exchange and LF is litterfall. Uptake of Bc and N is coupled to the mass balance (equation 7.113) by the respective production term r_y . In the *SAFE* model, all N -deposition is assumed to be, or to be converted to, nitrate and all N -uptake therefore generates alkalinity. It should be pointed out, that the reaction path that ammonium takes in the topmost layer is insignificant, as long as the key assumption regards no ammonium leaching from the topmost layer. Multi-layer models, such as the *SAFE* model, require uptake to be distributed to the different layers. In the *SAFE* model, the model user specifies the two distributions according to which N and Bc gross uptake should be distributed. As the *SAFE* model do not model vegetation growth and decomposition of organic matter, either the model user or a separate model have to keep track of the pool of nutrients in the organic matter and the pool of nutrients in the vegetation. If no net mineralisation is specified, the pool of half degraded litter on the forest floor and the pool of organically bound nutrients in the soil are at steady state. *SAFE* do not contain any checks to see if there is enough N and it is thus crucial that the specified N net uptake do not exceed what is available through deposition and net mineralisation. If uptake is reconstructed using the reconstruction method *MAKEDEP*, N -uptake already meets this condition since *MAKEDEP* assumes that the growth is N -limited with the maximum net uptake of N equal to N deposition. The current version of the *INITSAFE* model displays warnings if the N balance is violated. Bc limitations can not be considered in advance, since the Bc availability is dependent on the weathering rate calculated by *SAFE*. Bc uptake therefore becomes an issue about supply and demand on the level of *SAFE*. In situations with low Bc concentrations, the given base cation uptake in a soil horizon is decreased can be adjusted according to Eq 7.26.

7.7.2 Soil solution chemistry

The soil solution chemistry in SAFE and PROFILE is based on generic models where the pH-ANC relationship plays a prominent role. All the major ions are modelled, but several simplifications have been made. Buffering in the liquid phase is controlled by the CO₂-carbonate system, the acid-base reactions of a monovalent organic acid RH and reactions between differently charged Al-hydroxyl complexes. Due to the fast reactions involved, it is not feasible to calculate pH using a mass balance for H. pH, the negative logarithm of the H⁺ concentration, is therefore calculated using a mass balance for ANC, where ANC is defined by rearranging a charge balance either as

$$[\text{ANC}] = 2[\text{Ca}^{2+}] + 2[\text{Mg}^{2+}] + [\text{K}^+] + [\text{Na}^+] + [\text{NH}_4^+] - [\text{Cl}^-] - 2[\text{SO}_4^{2-}] - [\text{NO}_3^-]. \quad (7.122)$$

or, using ions involved in equilibrium reactions. Equilibrium is in this thesis used as a short for *chemical* equilibrium. At chemical equilibrium, the overall reaction rate is zero and the term equilibrium can thus only be used for reversible reactions.

$$[\text{ANC}] = [\text{OH}^-] + [\text{HCO}_3^-] + 2[\text{CO}_3^{2-}] + [\text{R}^-] - [\text{H}^+] - 3[\text{Al}^{3+}] - 2[\text{AlOH}^{2+}] - [\text{Al}(\text{OH})_2^+] \quad (7.123)$$

The species in the ANC definition in equation 7.123 can be linked to the H⁺ concentration through equilibrium equations for the auto-protolysis of water, the carbonate system, Henry's law for P_{CO₂}-H₂CO₃ equilibrium, equilibrium equations between Al-hydroxy species and finally a cubic curve fit, the gibbsite apparent gibbsite solubility, determining the Al concentration. To be strict, activities, e.g. *a*_{OH}, rather than concentrations, e.g. [OH], should be used. The effect of ionic strength is however ignored in the current versions of PROFILE and SAFE.

$$[\text{OH}^-] = \frac{K_W}{[\text{H}^+]} \quad (7.124)$$

$$[\text{HCO}_3^-] = \frac{K_{\text{H}_2\text{CO}_3} - K_{\text{Henry}} P_{\text{CO}_2}}{[\text{H}^+]} \quad (7.125)$$

$$[\text{CO}_3^{2-}] = [\text{HCO}_3^-] \frac{K_{\text{HCO}_3^-}}{[\text{H}^+]} \quad (7.126)$$

$$[\text{Al}^{3+}] = K_G [\text{H}^+]^3 \quad (7.127)$$

$$[\text{AlOH}^{2+}] = K_{\text{AlOH}^{2+}} K_G [\text{H}^+]^2 \quad (7.128)$$

$$[\text{Al}(\text{OH})_2^+] = K_{\text{Al}(\text{OH})_2^+} K_G [\text{H}^+] \quad (7.129)$$

where P_{CO₂} is the partial CO₂ pressure (atm) and *K_W* etc. are aqueous equilibrium coefficients as specified in Table 7.7.2. It should carefully be noted that the reason for selecting the apparent gibbsite solubility as the foundation for modelling aluminium is primarily that reasonable titration curves can be derived with this aquatic chemistry model, and *not* due to a conviction that gibbsite is the actual solid phase determining Al concentrations in terrestrial systems.

In the organic soil layers, the buffering reaction of organic components may also be important. Dissolved organic carbon (DOC) is modelled as a monovalent organic acid. The dissociation of the acid functional groups of the DOC is quantified using the Oliver equation

$$[\text{R}^-] = \frac{K_{\text{Oliv}} [\text{DOC}] \alpha}{K_{\text{Oliv}} + [\text{H}^+]} \quad (7.130)$$

| Equilibrium reaction | Eq. coeff. | Value/function | Ref. |
|--|--------------------------------|-------------------------------------|------|
| $\text{H}_2\text{O} \rightleftharpoons \text{H}_3\text{O}^+ + \text{OH}^-$ | K_W | $10^{(6.09-4471/T-0.0171T)}$ | 1 |
| $\text{CO}_2(g) \rightleftharpoons \text{H}_2\text{CO}_3(aq)$ | K_{Henry} | $10^{(-12.59+2198/T+0.0126T)}$ | 1 |
| $\text{HOC}_3^- + \text{H}_2\text{O} \rightleftharpoons \text{HCO}_3^- + \text{H}_3\text{O}^+$ | $K_{\text{H}_2\text{CO}_3^-}$ | $10^{(14.82-3401/T-0.0327T)}$ | 1 |
| $\text{HCO}_3^- + \text{H}_2\text{O} \rightleftharpoons \text{R}^- + \text{H}_3\text{O}^+$ | $K_{\text{HCO}_3^-}$ | $10^{(6.53-2906/T-0.0238T)}$ | 1 |
| $\text{RH} + \text{H}_2\text{O} \rightleftharpoons \text{Al}^{3+} + 6\text{H}_2\text{O}$ | K_{oliv} | $10^{-0.96-0.9pH+0.039 \cdot pH^2}$ | 2 |
| $\text{Al}(\text{OH})_3 + 3\text{H}_3\text{O}^+ \rightleftharpoons \text{Al}^{3+} + 6\text{H}_2\text{O}$ | K_G | specified in input | |
| $\text{Al}^{3+} + 4\text{H}_2\text{O} \rightleftharpoons \text{Al}(\text{OH})_2^+ + 2\text{H}_3\text{O}^+$ | $K_{\text{Al}(\text{OH})_2^+}$ | $5 \cdot 10^{-10}$ | 3 |
| $\text{Al}^{3+} + 2\text{H}_2\text{O} \rightleftharpoons \text{AlOH}^{2+} + \text{H}_3\text{O}^+$ | $K_{\text{AlOH}^{2+}}$ | $1 \cdot 10^{-5}$ | 3 |

Table 7.27: *Equilibrium constants used in the SAFE and PROFILE models. T is the temperature in Kelvin.*

where DOC is dissolved organic carbon in g m^{-3} , α a conversion factor ($\alpha = 7 \cdot 10^{-6} \text{ kmol g}^{-1}$) and K_{Oliv} as listed in Table 7.7.2. Since the solution H^+ concentrations determines the equilibrium concentration of all species in the definition of ANC (equation 7.123) and the resulting equation is not explicitly solvable with respect to H^+ concentration, pH must be calculated iteratively from ANC. ANC is often referred to as alkalinity if positive and as acidity if negative. It should be noted that operational definitions based on different measurement methods exist for alkalinity and acidity and that numerical value from such measurements might deviate slightly from ANC as defined above. As mentioned earlier, the SAFE and PROFILE models uses a mass balance (equation 7.113) to calculate ANC in each soil horizon. To check model consistency, ANC is also calculated using equation 7.122 and both values, which should be identical, are printed to the result files.

7.7.3 Cation exchange

Although the cation exchange process have been known for at least 150 years (Ruvarc 1993), different methods to describe the cation exchange process is still being investigated (Snyder and Cavallaro 1997; Morgan et al 1995; Hall 1996). In SAFE, cation exchange plays a major role for creating delays. Recently, a sulphate adsorption model has also been developed, but the extra cost of input data is seldom warranted by the rather marginal increase in model performance. It has for most applications been turned off and ignored.

Cation exchange molecular mechanisms

There are two thermodynamically different ways to formulate the cation exchange reaction, the Vanselow convention and the Gapon convention. While the Vanselow convention is based on the molar quantities of the exchanged ions, the Gapon convention is based on the equivalent quantities of the exchanged ions. The Vanselow exchange reaction is written as (Sposito 1977):



whereas the Gapon exchange reaction is written as:



where MX_m designates an ion M of charge $m+$ and all the m sites of type X to which it is bonded and $\text{M}_{1/m}\text{X}$ designates an ion M and one of the m sites of type X to which M is bonded. M

and N are thus cations of charge $m+$ and $n+$ respectively. To put it in other words, the Gapon convention can be said to focus on the individual bonds between the ions and the exchange surface rather than the ions themselves. Consequently, the Gapon and Vanselow conventions are only equivalent for the special case when $m = n = 1$. According to equation 7.132, the Gapon exchange equation is thus written as¹

$$k''_{Gapon} = \frac{\{M^{m+}\}^n \{N_{1/n}X\}^{nm}}{\{N^{n+}\}^m \{M_{1/m}X\}^{mn}} \quad (7.133)$$

or

$$k'_{Gapon} = \frac{\{M^{m+}\}^{\frac{1}{m}} \{N_{1/n}X\}}{\{N^{n+}\}^{\frac{1}{n}} \{M_{1/m}X\}} \quad (7.134)$$

where $\{\}$ denote activity. The Gapon convention further assumes the solute activities to equal the molar concentrations and the exchanger activities to equal the equivalent fractions which gives us

$$k'_{Gapon} = \frac{[M^{m+}]^{\frac{1}{m}} E_N}{[N^{n+}]^{\frac{1}{n}} E_M} \quad (7.135)$$

where E_N and E_M are the equivalent fractions² of ion N^{n+} and M^{m+} respectively on the cation exchange surface. For the Vanselow exchange equation,

$$K = \frac{\{M^{m+}\}^n \{NX\}^m}{\{N^{n+}\}^m \{MX\}^n}, \quad (7.136)$$

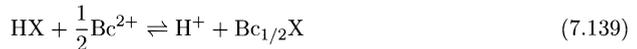
there are a number of different conventions for describing the activity at the exchange surface. Two common conventions are the Vanselow-Argersinger convention which is based on molar fractions and the Gaines-Thomas convention which is based on equivalent fractions (Bond 1995; Gaines and Thomas 1953). The conditional equilibrium constants for these two conventions can be written as (Bond 1995):

$$k_{Vanselow} = \frac{\{M^{m+}\}^n X_N^m}{\{N^{n+}\}^m X_M^n} \quad (7.137)$$

$$k_{Gaines-Thomas} = \frac{\{M^{m+}\}^n E_N^m}{\{N^{n+}\}^m E_M^n} \quad (7.138)$$

where X_M and X_N are the molar fractions of ion N^{n+} and M^{m+} respectively on the cation exchange surface. In the SAFE model, cation exchange is treated as a process including all processes where an adsorbed cation is exchanged with another/other cations. The Gapon cation exchange equation is used to calculate all surface exchange processes, independent of the actual mechanism. Surface complexation as well as diffuse layer adsorption is thus included in the cation exchange process.

In the SAFE model, three different species are considered to be adsorbed on the cation exchange surface, namely Al^{3+} , H^{3+} and Bc. Since Ca, Mg and K is lumped into one cation Bc, the difference in charge between K and the other cations can not be addressed: A lumped Bc ion can only have one charge. The system can therefore be described by two Gapon exchange reactions



¹This is simply an equilibrium equation using the law of mass action.

²Notation has changed as compared to previous SAFE model descriptions to comply with current cation exchange literature.

where E_Y is the fraction of sites on the exchange surface to which species Y is adsorbed. The exchange between Al^{3+} and H^+ does not need to be written explicitly, since it is given by the sum of the two reactions 7.139 and 7.140. The system can thus be described by two Gapon exchange equations one additional constraint

$$k'_{H/Bc} = \frac{[H^+]}{[Bc^{2+}]^{\frac{1}{2}}} \frac{E_{Bc}}{E_H} \quad (7.141)$$

$$k'_{H/Al} = \frac{[H^+]}{[Al^{3+}]^{\frac{1}{3}}} \frac{E_{Al}}{E_H} \quad (7.142)$$

$$E_{Bc} + E_{Al} + E_H = 1. \quad (7.143)$$

If we assume, as discussed earlier, that the Al^{3+} concentration is related to the H concentration through an apparent gibbsite coefficient

$$[Al^{3+}] = K_G [H^+]^3, \quad (7.144)$$

the H/Al exchange (equation 7.142) can be transformed to

$$k'_{H/Al} K_G^{\frac{1}{3}} = \frac{E_{Al}}{E_H}. \quad (7.145)$$

Combining equation 7.145 with equation 7.143 result in

$$E_H = \frac{1 - E_{Bc}}{1 + K_G^{\frac{1}{3}} k'_{H/Al}} \quad (7.146)$$

which in combination with the H/Bc exchange equation (equation 7.141) yields

$$\frac{k'_{H/Bc}}{\left(1 + K_G^{\frac{1}{3}} k'_{H/Al}\right)} = \frac{[H^+]}{[Bc^{2+}]^{\frac{1}{2}}} \frac{E_{Bc}}{(1 - E_{Bc})}. \quad (7.147)$$

Thus, only one exchange equation,

$$k_{H/Bc} = \frac{[H^+]}{[Bc^{2+}]^{\frac{1}{2}}} \frac{E_{Bc}}{(1 - E_{Bc})} \quad (7.148)$$

is needed to describe the exchange of Bc, as long as the relation between the Al^{3+} and H^+ concentrations at the exchange surface is given by an apparent gibbsite coefficient. If needed, the equivalent fractions E_H and E_{Al} can be calculated directly from E_{Bc} (equation 7.145 and 7.143). It should be noted that the E_{Al} to E_H ratio will be constant if the concentrations at the surface can be described by a cubic relationship between Al and H (equation 7.145). Changes in the E_{Al} to E_H ratio reported in the literature (Matzner 1988) indicate problems with the cubic relationship assumption.

Diffusion-limited cation exchange in SAFE

Traditionally, cation-exchange has been modelled as an equilibrium reaction, which also is the case in the MAGIC and SMART models. Given the episodic variations in water flux and soil solution concentrations of involved cations, mass-transfer between the bulk solution and the exchange surface most likely limits the cation exchange rate. As the intrinsic rate for the cation exchange reaction is considered to be infinite, equilibrium can still be assumed between the conditions on the exchange surface and the solution concentrations *at* the surface. Using Fick's

first law and a mass-balance over the surface and the diffusion layer, the differential equation for the conditions at the surface can be written as (Warfvinge and Sverdrup 1988)

$$\frac{dE_{\text{Bc}}}{dt} = 2k_x ([\text{Bc}^{2+}] - [\text{Bc}^{2+}]_{\text{surface}}) \quad (7.149)$$

where subscript surface indicates that the conditions at the exchange surface are considered and where k_x is the mass transfer coefficient. The flux of Bc to the exchange surface is thus proportional to the concentration gradient between the solution and the surface. Combining equation 7.149 with the mass balance for base saturation (equation 7.120) yields

$$r_{E_{\text{Bc}}} = k_x z \rho \text{CEC} ([\text{Bc}^{2+}]_{\text{surface}} - [\text{Bc}^{2+}]) \quad (7.150)$$

The Bc concentration at the exchange surface is still given by the Gapon equation (equation 7.148)

$$[\text{Bc}^{2+}]_{\text{surface}} = \frac{[\text{H}^+]_{\text{surface}}^2 E_{\text{Bc}}^2}{k_{\text{H/Bc}}^2 (1 - E_{\text{Bc}})^2} \quad (7.151)$$

where the H concentration at the exchange surface, $[\text{H}^+]_{\text{surface}}$ is in an implicit function of ANC at the surface. As discussed earlier (equation 7.122), ANC can be expressed as

$$\text{ANC} = 2[\text{Bc}^{2+}] + [\text{Na}^+] + [\text{NH}_4^+] - [\text{NO}_3^-] - 2[\text{SO}_4^{2-}] - [\text{Cl}^-] \quad (7.152)$$

and since Bc is the only ion in the above equation subject to cation exchange, the relation between ANC at the surface and the ANC in the bulk solution is given by

$$\text{ANC} - 2[\text{Bc}^{2+}] = \text{ANC}_{\text{surface}} - 2[\text{Bc}^{2+}]_{\text{surface}}. \quad (7.153)$$

This relation combined with the Gapon equation yields³

$$[\text{H}^+]_{\text{surface}} = \frac{1 - E_{\text{Bc}}}{E_{\text{Bc}}} k_{\text{H/Bc}} \sqrt{[\text{Bc}^{2+}] - (\text{ANC} - \text{ANC}_{\text{surface}})/2}, \quad (7.154)$$

which is solved by a bisection iteration of $[\text{H}^+]_{\text{surface}}$ for a given ANC.

Running SAFE

Initial conditions, PROFILE and INITSAFE There are at least two reasons for starting dynamic simulations of forest ecosystems from pristine conditions:

1. Starting from a steady state assures that the model output is determined by changes in the external and internal load, rather than by transients due to unstable initial conditions. To be able to assume steady-state starting conditions, it is necessary to start the simulations before acidification sets in.
2. Measurement, e.g. of deposition and soil solution chemistry, is usually only available at a single point in time. It is therefore usually not possible to validate the dynamic behaviour in applications of soil chemistry models. Consequently, if the model can not reconstruct reasonable dynamics for the past, we have no reason to believe model forecasts of future dynamics. Starting from pristine conditions, thus increases the transparency and reliability of the model application.

³In previous model descriptions and in early versions of the SAFE model this equation contained a minor error

The PROFILE and INITSAFE models are steady state versions of SAFE, which bypasses the changes in soil status over time and calculates the final steady state directly. Three different varieties of PROFILE exists, the single-site version with a user-friendly interface, a regional version for calculating weathering and B/Al ratios, and a regional version for calculating critical loads. PROFILE and SAFE have evolved certain differences over time. As we recommend SAFE to be started from stable initial conditions, the INITSAFE model should be used rather than the PROFILE model calculate the initial state as the INITSAFE model contains exactly the same process descriptions as the SAFE model. In PROFILE and SAFE the differentials in the mass balances are zero, and the generic mass balance (equation 7.113) thus becomes

$$0 = \frac{1}{z\theta} (Q_0[Y]_0 - Q[Y]) + r_Y . \quad (7.155)$$

The mass balance in the INITSAFE model are otherwise identical with those of the SAFE model except, of course, for the omitted mass balance for base saturation since the base saturation is constant at a steady-state. In the single-site version of the PROFILE model, NO_3^- and NH_4^+ are treated separately using a nitrification rate constant and Ca^{2+} , Mg^{2+} and K^+ are treated separately rather than as a lumped cation.

Model calibration In the SAFE model, the underlying principle is that calculated base saturations and soil solution chemistry should be directly measurable in field and possibilities to calibrate the model should be limited. The SAFE model is a multi-layer model rather than a one-layer model. The soil should ideally be divided into layers according to the naturally occurring soil horizons. The base saturation and soil solution chemistry calculated in each layer should therefore be compared with the conditions in the corresponding soil horizons. Weathering rate is, as already mentioned, calculated in a sub-model. The weathering rates calculated by SAFE thus reflect the mineralogy of the soil and the elemental composition of the minerals. Starting the SAFE model from stable initial conditions also precludes the possibility to calibrate model output by tampering with the initial conditions. To further limit calibration of the SAFE model, only calibration on base saturation is implemented. Although it is possible to calibrate SAFE model output to a certain extent by changing the distribution of nutrient uptake and the apparent gibbsite solubility coefficients all such calibrations intentionally have to be made manually in order to restrain such calibrations.

The SAFE model is calibrated by adjusting the initial base saturation in order to achieve maximum agreement between recent measured and modelled base saturation. Each layer is calibrated separately, but the lower layers are of course affected by the chemistry of layers above. The initial base saturation is calibrated until

$$\left| \sum_i (E_{\text{Bc, simulated}_i} - E_{\text{Bc, measured}_i}) \right| \leq \epsilon \quad (7.156)$$

where $E_{\text{Bc, measured}_i}$ is the i :th measurement of base saturation, $E_{\text{Bc, simulated}_i}$ the corresponding simulated base saturation and ϵ the accepted error. As initial soil solution variables (ANC, base cation concentration, pH, etc) are calculated independently assuming steady-state, the possibilities of additionally "tuning" SAFE are relatively few. Therefore, shortcomings in the model formulation are likely to show up in model/data comparisons. The Al concentration in the SAFE model is controlled by the gibbsite model. The pK_{Gibb} used had the values systematically used in mapping critical loads (Sverdrup et al., 1990); 6.5, 7.5, 8.5, 9.5. The gibbsite model is obviously an oversimplification of reality.

| Parameter | Unit | Site | Regional |
|-----------------------------------|--------------------------------------|---------------------------|----------------|
| Morphological characterization | | From data | Classification |
| Soil layer thickness | m | From data | Classification |
| Moisture content | $\text{m}^3 \text{m}^{-3}$ | From data | Classification |
| Soil bulk density | kg m^{-3} | From data | Classification |
| Specific surface area | $\text{m}^2 \text{m}^{-3} * 10^{-6}$ | From data | From data |
| Inflow | % of precipitation | From data | From data |
| Cation exchange capacity (CEC) | $\text{keq kg}^{-1} \cdot 10^{-6}$ | From data | From data |
| Base saturation, calibration year | % | From data | From data |
| K-feldspar | % of total | From data | UPPSALA model |
| Oligoclase | % of total | From data | UPPSALA model |
| Albite | % of total | From data | UPPSALA model |
| Hornblende | % of total | From data | UPPSALA model |
| Pyroxene | % of total | From data | UPPSALA model |
| Epidote | % of total | From data | UPPSALA model |
| Garnet | % of total | From data | UPPSALA model |
| Biotite | % of total | From data | UPPSALA model |
| Muscovite | % of total | From data | UPPSALA model |
| Chlorite | % of total | From data | UPPSALA model |
| Vermiculite 1 | % of total | From data | UPPSALA model |
| Vermiculite 2 | % of total | From data | UPPSALA model |
| Vermiculite 3 | % of total | From data | UPPSALA model |
| Illite 1 | % of total | From data | From data |
| Illite 2 | % of total | From data | From data |
| Illite 3 | % of total | From data | From data |
| Smectite | % of total | From data | From data |
| Apatite | % of total | From data | UPPSALA model |
| Calcite | % of total | From data | From data |
| Deposition mass fluxes | keq/ha yr | data/MAKEDEP | data/MAKEDEP |
| Initial pH | | From InitSAFE | From InitSAFE |
| Initial BC concentration | $\mu\text{mol}(+) \text{m}^{-3}$ | InitSAFE | InitSAFE |
| CO ₂ pressure | times ambient | From data/DECOMP | Classification |
| Dissolved organic carbon | mg l^{-1} | data/DECOMP | Classification |
| Precipitation fluxes | mm of rain | data/GROW | From data |
| Evapotranspiration | % of precipitation | FLOW | From data |
| Mg+Ca+K uptake | % of total max | GROW | data/GROW |
| N uptake | % of total max | From GROW | data/GROW |
| Al equilibrium constant | $\text{kmol}^2 \text{m}^{-3}$ | Default values | Default values |
| Mg+Ca+K uptake distribution | % of total | Default root distribution | Classification |
| N uptake distribution | % of total | Default root distribution | Classification |

Table 7.28: PROFILE and SAFE input data. The difference in input between PROFILE and SAFE is that SAFE also requires CEC, initial [BC], [Al] and BS for calibration and the fact that SAFE requires input as time-series.

7.8 Weathering of soil minerals

Several different models or methods have been developed to estimate soil mineral weathering. However, in this report we will focus on the soil model PROFILE developed by Sverdrup and Warfvinge (1993, 1995). PROFILE is a mathematical, biogeochemical steady state model and was developed with the objective to calculate the soil solution chemistry and to estimate the effects of acid deposition on soils and groundwater (Fig. 7.43). In PROFILE the soil is stratified in a series of mixed compartments, where each compartment represent different soil layer. Each soil layer has an individual character and is assumed to be well mixed with no internal chemical gradient. Soil stratification in PROFILE enhances structural reliability but involves increased data needs. The PROFILE model contains several chemical subsystems that are reviewed and evaluated continuously, even new subsystems are developed (see previous sections) that can be fitted in to existing soil models. The main subsystems can be summarized as;

- Deposition, leaching and accumulation of dissolved chemical components
- Chemical weathering reactions of soil mineral
- The net result of soil reactions with nitrogen compound
- Internal cycling of nitrogen and base cations in the canopy, such as uptake, canopy exchange, litter fall and mineralization
- Net uptake of base cations and nitrogen removed by forest management
- Solution equilibrium reactions involving the carbonate system, CO_2 , speciation of aluminium and organic acids

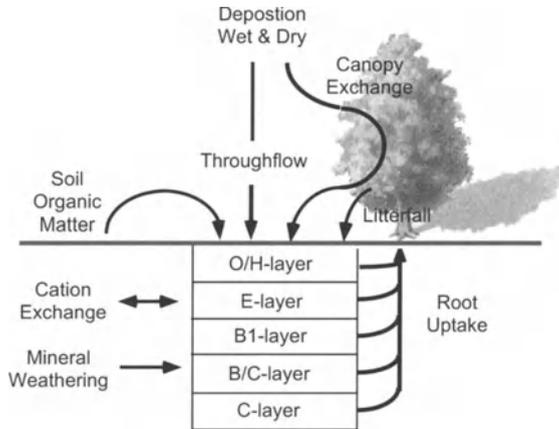


Figure 7.43: A schematic figure of the components and biogeochemical cycles that are built in the PROFILE model.

Other sub-models that will be included in the PROFILE in future are (excluded the ones that have been mentioned in previously sections):

- Internal cycling and decomposition of organic carbon
- Leaching and accumulation of heavy metals

The main chemical processes such as nitrification, denitrification and weathering are modeled as temperature dependent processes. The weathering of soil minerals is dependent on the soil environment like hydrogen and cation concentration, and organic acids. Therefore, is the model accounting for the different chemical variables in the soil solution when the weathering rate is calculated. As stated before is PROFILE a multi-layer model and allows therefore to be parameterized for all individual soil horizons. On the other side is PROFILE regarded as quite

parameter intensive and need an extensive set of input data to minimize the uncertainty. The input data for PROFILE may be subdivided into groups and comprise in short:

Climate Physical climate; Mean temperature, annual precipitation and runoff. Chemical; Atmospheric deposition of sulphate, nitrate, chloride, ammonium, sodium, calcium, magnesium and potassium

Vegetation Tree species and their distribution, annual base cations and nitrogen uptake, base cations and nitrogen in litter fall

Soil Physics; Exposed mineral surface area, moisture, bulk density, soil layer thickness, Geology; mineralogy Chemistry; Carbon dioxide pressure, soil solution DOC, Al solubility

The geochemical study in the Asa Forest Research Park has mainly been focused at sampling and analyze the forest soil to determine the soil type and other chemical and physical soil parameters. The sampling strategy has been to get sufficient data creating a statistically sound map for the weathering for the whole of the western part Asa Research Park, and area of approximately 1,800 ha. The sampling density and spatial pattern was chosen to allow us to perform a geostatistical analysis of the map and make a kriged surface for the area, equivalent to a continuous map for weathering rate.

7.8.1 Weathering and the PROFILE model

It is important to realize that it is not possible to assess single soil processes without taking all other significant processes quantitatively into account. PROFILE and its time variant counterpart SAFE take into account simultaneously the chemical weathering rate, base cation, ammonium and nitrate uptake by the vegetation, nitrification of ammonium to nitrate, as well as solution reactions involving the carbonate system, aluminium species and organic acids and let these processes interact. The deposition of sulphate, ammonium, nitrate and base cations is considered. These processes are represented by mass balances and kinetic equations inside the PROFILE model. The model operates on mass balance equations for ANC, base cation, nitrate and ammonium. The interactions between the phases and the reactants in the system always take the path via the soil solution. Other types of soil chemistry models available, which treat the silicate weathering component either as a calibrated input parameter or as a process at equilibrium. Equilibrium assumptions as the basic principle for irreversible weathering reactions, deserves no further mention.

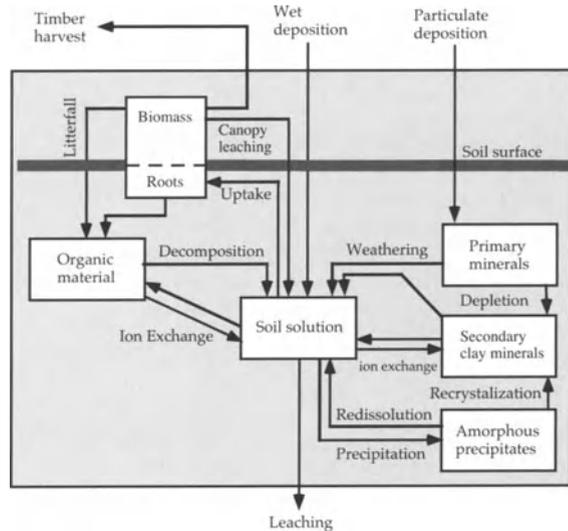


Figure 7.44: The figure show the internal structure of the revised PROFILE model. All weathering process must considered in the context of other important soil processes, and ignoring to do so inevitably leads to poor performance the model is tested under field conditions. The diagram above show the pathways of base cations.

Principles of the model platform

The Swedish PROFILE soil chemistry model has a substantial submodel for weathering, based on first principles. The weathering rate model is well adapted and validated for Scandinavian soils from glaciated areas, soils rich in primary minerals and with small amounts of secondary minerals.

The PROFILE has been used as a point of departure for creating a soil weathering model for soils dominated by clay minerals. During the work associated with determining weathering rates for forest soils in Germany, it was realized that it was necessary to adapt and test the PROFILE model to the lowland soils of the central European plains. These soils are rich in clay minerals, have not been glaciated in the last million years and they have very low contents of primary minerals. PROFILE is a steady state model, which implies that all time dependent processes have been short-circuited. It is closely related to the dynamic model SAFE. It is necessary to consider the interactions of the weathering process with other soil processes when laboratory experiments are to be related to field observations. Laboratory kinetic coefficients for primary minerals have been integrated into SAFE and PROFILE earlier. The general model includes the following chemical subsystems: (1) Deposition, leaching and accumulation of the dissolved chemical components, (2) Chemical weathering, (3) Cation exchange, (4) Nitrification and denitrification, (5) Cycling of nitrogen and base cation between the vegetation and soil, (6) Net uptake of base cations and nitrogen and (7) Solution equilibrium reactions. SAFE operates on a variable timestep, downward adjustable to hourly steps, and weathering rates at Swedish research sites have been calculated with hourly resolution. The model is structured into a series of layers, each modeled as a stirred tank. The mass balances are solved for each soil layer, solute equilibria, kinetic rates and weathering. Input data are assembled for the model according to layers. A number of weathering reactions have been experimentally identified to take place in parallel at the mineral surface (Sverdrup 1990). Simultaneous reaction systems are considered where the mineral surface reacts with (1) hydrogen ions, (2) water, (3) hydroxyl ions, (4) carbon dioxide and (5) organic acids ligands. The total base cation release rate by chemical weathering will be the sum of the rate of all parallel simultaneous processes regardless of the molecular mechanism, minus the rate of base cation co-precipitation with secondary solid phases. We can write for the forward rate:

$$r_W = r_{H^+} + r_{H_2O} + r_{OH^-} + r_{CO_2} + r_{Org} \quad (7.157)$$

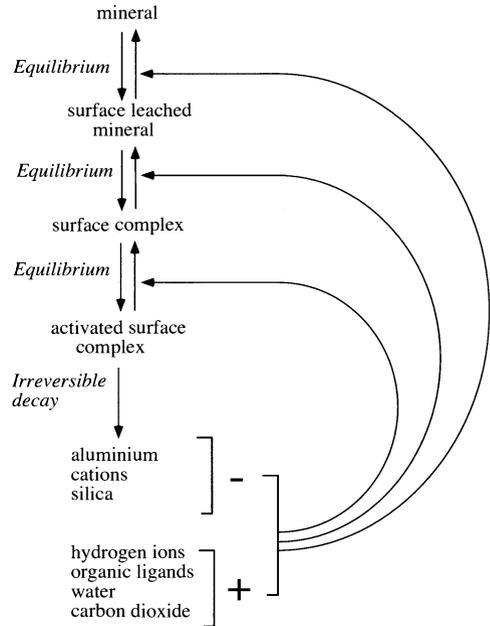


Figure 7.45: Simple causal loop diagram for the soil mineral weathering process. Fundamental is the occurrence of an irreversible step from activated surface complex to the dissolved constituents. This effectively rules out any equilibrium approach as a flagrant breach of the second law of thermodynamics. For clay minerals, changes in the fundamental chemical conditions, may cause some of the reversible steps to become slower than the irreversible step or to temporarily change the direction of the overall reaction.

Studies over many years on four dozen different minerals have shown that Eq. 5.168 *always* apply, and ignoring this in the experimental design makes it likely that the results are not useful. The initial stage of mineral dissolution is also characterized by non-stoichiometric dissolution where the molar composition of the reaction products is different from the molar composition of the parent mineral. During long term steady state dissolution, most mineral dissolution is stoichiometric, and any deviation from this, usually indicates that some secondary solid phase is simultaneously precipitated from the solution. The rate depends on the amount of mineral surface available, which determines the amount of activated surface complexes that can participate in the process. From the reaction, a solid residual is produced (Fig. 7.45). The rate is known to be dependent on the hydrogen ion concentration of the solution with a power between 0.5 and 0.7. However, there is more to it than this. Experiments have shown that the aluminium concentration has a strong retarding effect on the rate (Sverdrup 1990), and this must be included in the picture (Fig. 7.46). The same experiments show that base cations and silica both affect the rate negatively and the formation of precipitates (Sverdrup 1990). Thus the rate expression for each reaction must include the effect of reactant promotion as well as the retarding feedback from reaction products. Further the rate equation for mineral dissolution must include the sum of rates of all the reactions involved. We can safely state that experiments or models that does not consider the four significant and simultaneous reactions and at the same time the significant interactive feedbacks, they will quickly run into well known interpretation difficulties and fail to be useful. For each of the reactions included in Eq. 7.157, a rate expression based on the transition state theory may be derived (Sverdrup 1990). Under certain circumstances, sheet silicates may dissolve partially or non-stoichiometrically, and form secondary mineral residues. For each of the reactions included in Eq. 7.157, a rate expression based on the transition state theory may be derived (Sverdrup 1990). The kinetic equation applied to both the experiments and the PROFILE model is exemplified by the expression for the specific reaction between the hydrogen ion and the mineral surface;

$$r_i = k_{H^+} \cdot \frac{a_{H^+}^n}{a_{BC}^x \cdot a_{Al^{3+}}^y} + k_{H_2O} \cdot \frac{a_{H_2O}}{a_{BC}^z \cdot a_{Al^{3+}}^w} + k_{CO_2} \cdot \dots + \dots \quad (7.158)$$

r is the reaction rate ($\text{keq m}^{-2} \text{s}^{-1}$), k_{H^+} the rate coefficient in the H^+ reaction. a_{Al} , a_{BC} and a_{H^+} are the activities of aluminium, base cation and hydrogen ion activities. x , z , y , w are reaction orders. It can be seen that a lot of constituents participate, and this must be considered in the experimental design. The total weathering rate for the soil is obtained by repeating the calculation additively for all minerals present:

$$R = \sum_{j=1}^{minerals} r_j \cdot A_j \quad (7.159)$$

R is the cumulative weathering rate in the soil ($\text{keq ha}^{-1} \text{yr}^{-1}$). The rate coefficients in the kinetic expression are all subject to change with temperature. The reactions only will take place on wetted surfaces, and the degree of surface wetting is taken to be proportional to the soil moisture saturation. All surfaces to participate in the reactions must be wetted, but there must also be sufficient soil solution present for the weathering process to respond to and exchange mass with other soil processes. The active surface area is adjusted for this. Mere adsorption of molecular water to the surface is not enough, the mineral must be wetted and stay in contact with other soil particles by means of a solution, hence the filling up approach. In PROFILE and SAFE models, the implemented rate laws for the release of the base cations Ca, Mg, K and Na (BC) from weathering of silicate minerals are based on a synthesis of transition state theory and data from field and laboratory measurements (Sverdrup 1990);

$$r_j = k_{H^+} \frac{[H^+]^{n_{H^+}}}{f_{H^+}} + k_{H_2O} \frac{1}{f_{H_2O}} + k_{CO_2} \frac{P_{CO_2}^{n_{CO_2}}}{f_{CO_2}} + k_R \frac{[R^-]^{n_{R^-}}}{f_{R^-}} \quad (7.160)$$

where r_j is the dissolution rate of mineral j in $\text{kmol}_c \text{ m}^{-2} \text{ s}^{-1}$, n are reaction orders, k_{H^+} the rate coefficient for the H^+ reaction, unit dependent on n_{H^+} , $k_{\text{H}_2\text{O}}$ the rate coefficient for the H_2O reaction in $\text{kmol}_c \text{ m}^{-2} \text{ s}^{-1}$, k_{CO_2} the rate coefficient for the CO_2 reaction, unit dependent on n_{CO_2} , k_{R} the rate coefficient for the lumped organic acid reactions, unit dependent on n_{R^-} , P_{CO_2} the partial CO_2 pressure in the soil solution in atm and f are product inhibition factors. Each weathering reaction is product inhibited with product inhibition factors f , given by

$$f_{\text{H}^+} = \left(1 + \frac{[\text{Al}^{3+}]}{c_{\text{Al}}}\right)^{v_{\text{Al}}} \left(1 + \frac{[\text{Bc}^{2+}]}{c_{\text{Bc}}}\right)^{x_{\text{Bc}}} \quad (7.161)$$

$$f_{\text{H}_2\text{O}} = \left(1 + \frac{[\text{Al}^{3+}]}{c_{\text{Al}}}\right)^{z_{\text{Al}}} \left(1 + \frac{[\text{Bc}^{2+}]}{c_{\text{Bc}}}\right)^{z_{\text{Bc}}} \quad (7.162)$$

$$f_{\text{CO}_2} = \left(1 + \frac{[\text{Bc}^{2+}]}{c_{\text{Bc}}}\right)^{z_{\text{Bc}}} \quad (7.163)$$

$$f_{\text{R}^-} = \left(1 + \frac{[\text{R}^-]}{c_{\text{R}}}\right) \left(1 + \frac{[\text{Bc}^{2+}]}{c_{\text{Bc}}}\right)^{z_{\text{Bc}}} \quad (7.164)$$

where c_{Al} is the aluminium saturation constant in kmol m^{-3} , c_{Bc} is the base cation saturation constant in kmol m^{-3} and v, x, z are reaction orders. The weathering rate is thus dependent on pH and CO_2 pressure as well as on the concentrations of Bc, Al^{3+} and R^- . It is therefore not obvious how e.g. a decrease in Bc deposition with the associated increase in pH affect the weathering rate. Since the weathering reactions are temperature dependent, rate coefficients are adjusted to field temperature using an Arrhenius relationship

$$k = k_0 \cdot e^{-\frac{E_A}{RT}} \quad (7.165)$$

where E_A is the Arrhenius activation energy in J kmol^{-1} , R is the Universal gas constant in $\text{J kmol}^{-1} \text{ K}^{-1}$, T is the absolute temperature in K and k is the rate coefficient of a specific reaction. In order to relate the rate coefficient at one temperature with the rate coefficient at another, the Arrhenius relationship is transformed to

$$\log_{10} \left(\frac{k_1}{k_2}\right) = \frac{E_A}{R \ln 10} \left(\frac{1}{T_2} - \frac{1}{T_1}\right). \quad (7.166)$$

Rate coefficients, reaction orders and product inhibition limit concentrations for the H^+ , H_2O , CO_2 , and organic acid reactions for the minerals included in PROFILE and SAFE are supplied in a separate text file, mineraldata, which is read at runtime (appendix B). A user interface called Mineralstack is also available that facilitates reading and changing of the mineraldata file. Both the SAFE and PROFILE model assume the potential supply of each mineral to be infinite and the total surface area A_w and the surface area fractions x_j are therefore treated as constants. In a special version of PROFILE, however, attempts have been made to simulate changes in mineralogy since glaciation using mass balances for the supply of each mineral in each soil horizon. The total release rate of a base cation M^{m+} for one soil horizon is obtained by multiplying the weathering rates for all minerals present with their respective molar M content and to add it all up:

$$W_M = \sum_{j=1}^{\text{mineral}} r_j A_w x_j \Theta z \frac{y_{j,M}}{m} \quad (7.167)$$

where W_M is the release of M^{m+} in $\text{kmol m}^{-2} \text{ s}^{-1}$, A_w the exposed surface area of soil minerals in $\text{m}^2 \text{ m}^{-3}$, x_j is the surface area fraction of soil mineral j , Θ is the soil moisture saturation and $y_{j,M}$ the molar content of cation M^{m+} in mineral j . The reactions only will take place on

| Mineral | pk_H | n_H | C_{Al} | y_{Al} | C_{BC} | x_{BC} | pk_{H_2O} | z_{Al} | z_{BC} | pk_{CO_2} | n_{CO_2} | pk_{org} | C_R |
|----------------|--------|-------|----------|----------|----------|----------|-------------|----------|----------|-------------|------------|------------|-------|
| K-Feldspar | 14.7 | 0.5 | 4 | 0.4 | 500 | 0.15 | 17.5 | 0.14 | 0.15 | 16.8 | 0.6 | 15.0 | (5) |
| Plagioclase | 14.6 | 0.5 | 4 | 0.4 | 500 | 0.2 | 17.2 | 0.14 | 0.15 | 15.9 | 0.6 | 14.7 | (5) |
| Albite | 14.5 | 0.5 | 4 | 0.4 | 500 | 0.2 | 16.7 | 0.14 | 0.15 | 15.9 | 0.6 | 14.7 | (5) |
| Hornblende | 13.3 | 0.7 | 30 | 0.4 | 200 | 0.3 | 15.9 | 0.3 | 0.3 | (15.9) | (0.6) | 14.4 | (5) |
| Pyroxene | 12.3 | 0.7 | 500 | 0.2 | 200 | 0.3 | 17.5 | 0.1 | 0.3 | 15.8 | 0.6 | 14.4 | (5) |
| Epidote | 14.0 | 0.5 | 500 | 0.3 | 200 | 0.2 | 17.7 | 0.2 | 0.2 | (16.2) | 0.6 | (14.4) | (5) |
| Garnet | 12.4 | 1.0 | 300 | 0.4 | 500 | 0.2 | 16.9 | 0.2 | 0.2 | (15.8) | (0.6) | (14.7) | (50) |
| Biotite | 14.8 | 0.6 | 10 | 0.3 | (500) | 0.2 | 16.7 | 0.2 | 0.2 | 15.8 | (0.5) | 14.8 | (50) |
| Muscovite | 15.2 | 0.5 | 4 | 0.4 | (500) | 0.1 | 17.5 | 0.2 | 0.1 | 16.5 | (0.5) | 15.3 | (5) |
| Mg-Chlorite | 14.8 | 0.7 | (50) | (0.2) | (200) | (0.2) | (17.0) | (0.1) | (0.1) | 16.2 | (0.5) | 15.0 | (5) |
| Fe-Chlorite | 14.3 | 0.7 | (50) | (0.2) | (200) | (0.2) | (16.7) | (0.1) | (0.1) | 15.8 | (0.5) | 14.5 | (5) |
| Mg-Vermiculite | 15.2 | 0.6 | 4 | 0.4 | 500 | 0.2 | 17.6 | 0.1 | (0.1) | (16.5) | (0.5) | (15.6) | (5) |
| Fe-Vermiculite | 14.8 | 0.6 | 4 | 0.4 | 500 | 0.2 | 17.2 | 0.1 | (0.1) | (16.2) | (0.5) | (15.2) | (5) |
| Apatite | 12.8 | 0.7 | 100 | - | 300 | 0.4 | 15.8 | - | 0.2 | 15.8 | 0.6 | (19.5) | (5) |
| Kaolinite | 15.1 | 0.7 | 4 | 0.4 | 500 | 0.4 | 17.6 | 0.2 | 0.2 | (16.5) | (0.5) | 19.5 | (5) |
| Calcite | 13.6 | 1.0 | 5000 | 0.4 | 1000 | 0.4 | 15.2 | - | 0.2 | 13.2 | 1.0 | 13.2 | (5) |

Table 7.29: Approximate laboratory rate coefficients for the chemical weathering rate for minerals expressed as the flux of base cations (Ca, Mg, Na, K) related to a total rate expressed as keq/m^2s at $8^\circ C$, applied in the PROFILE model. Values within brackets are approximations. (Sverdrup and Warfvinge, 1995)

wetted surfaces, and the degree of surface wetting is taken to be proportional to the soil moisture saturation. All surfaces to participate in the reactions must be wetted, but there must also be sufficient soil solution present for the weathering process to respond to and exchange mass with other soil processes. The soil moisture saturation is calculated as the degree of filling up the soil void volume, by combining the densities of the solid, water, air and the bulk soil density, with the volumetric water content (Warfvinge and Sverdrup 1991; Sverdrup and Warfvinge 1993, 1995).

As Ca, Mg and K is modelled as a lumped Bc cation on equivalent basis, W_{Bc} is calculated as

$$W_{Bc} = W_{Ca} + W_{Mg} + \frac{W_K}{2} \quad (7.168)$$

The dependence of the total weathering rate on the soil moisture saturation Θ have two reasons. Firstly, only wetted surfaces are subject to chemical weathering. Secondly, water is needed as a medium to transport ions to and from the mineral surface, thus making the released cations available to nutrient uptake and other soil processes. At lower degrees of soil moisture saturation it would therefore seem reasonable to expect the reactivity of the exposed mineral surface to decrease (Sverdrup and Warfvinge 1987; Warfvinge 1988, Sverdrup 1990). The soil moisture saturation, the ratio between the water volume and void space is calculated by assuming

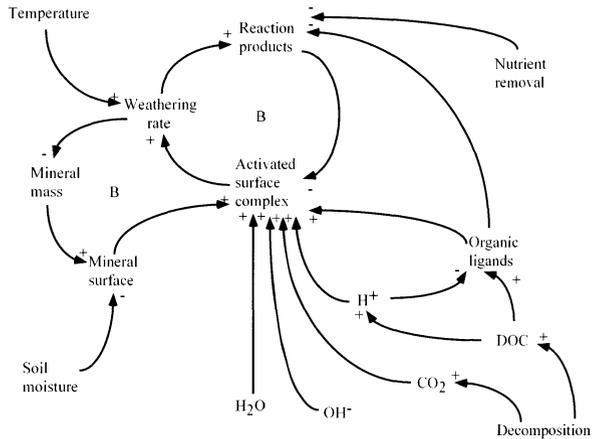


Figure 7.46: Simple causal loop diagram for the chemical weathering process. The weathering rate is strongly determined by the area of reactive surfaces made available to the soil water. Hydrogen ions attack the mineral and dissolve it. Experiments have shown that reaction products such as the Al and base cation released in the weathering reactions affects the dissolution rate, a phenomenon called product inhibition.

the soil to consist of three phases: solids, i.e. mineral grains and organic matter, water and gas. Since the mass of gas is negligible, the mass of the soil is equal to the sum of the solid mass and the water mass.

The PROFILE model has been designed as an "open" model, where the user may actually change many of the basic principles inside without having to recompile the whole model or touch the FORTRAN code. Principally, everything in the model is available from the authors, free of cost, including the source code. It is controlled from a set of cards. Fig. 7.47 shows the input data cards for PROFILE. For each layer, mineralogy, mineral surface area as reflected by texture, soil moisture etc. must be entered. The model has a separate stack of 14 cards for controlling the kinetics of mineral weathering. For each mineral there is one input card. This can be used to assign new kinetic coefficients, reaction orders, mineral stoichiometries and temperature dependencies to the model. This way new kinetic information can be integrated by any user, without recompilation of the whole model code. The model does not demand stoichiometric dissolution, the stoichiometry specifies the stoichiometry of the solid that dissolves, which may differ from the bulk composition of a particular mineral. The outputs from PROFILE are presented on a number of cards, Figs. 7.48. They show the amount of ions produced by chemical weathering, layer by layer, and ion by ion. The ions produced by the minerals considered are Ca, Mg, K, Na, Al, Si and P. The amounts of Al and Si are the total amounts produced by congruent dissolution at the mineral surface and not what appears in the soil solution. The PROFILE model was originally designed for estimating the sensitivity of soils to acid deposition. The weathering rate is one of the most important parameters for the acidification sensitivity, and accordingly, it was important to estimate it from geochemical and physical input data. The capability to predict field weathering rates arose from a systematical and specific way of interpreting the laboratory kinetics, and by incorporating the results in an integrated soil model, PROFILE (Sverdrup and Warfvinge 1987, 1988, 1993, 1995, Sverdrup 1990). The basis for making use of laboratory experiments for modeling, is to evaluate them in terms of a specific and consistent theory. The model permits data from experiments of very different designs to be used together in such a way that the effects of differences in conditions and properties can be accounted for, very different experiments can be normalized to a common platform for comparison. The basic principles applied in the PROFILE model has been developed in a series of causal loop diagrams (Figs. 7.45-7.46). For clay minerals, changes in the fundamental chemical conditions, may cause some of the reversible steps to become slower than the the irreversible step or to temporarily change the direction of the overall reaction.

The figure displays three screenshots of the PROFILE 4.1 software interface, showing input data cards for soil parameters, vegetation input, and mineralogy.

Top Screenshot: Profile 4.1 - General Parameters

| | | |
|--|--|---|
| Profile 4.1 | Clear | Go To Card |
| Defn Folder: [DataBank] | Save | Check Input |
| Run ID: [Gårdsfön P] | Import Data | Run Profile |
| Number of soil layers: 4 | <input type="checkbox"/> Add Results to file after run | <input type="checkbox"/> Do Streams Water |
| Precipitation rate (mm/yr): 0.9 | <input type="checkbox"/> Save Results to result bank. | |
| Runoff rate (mm/yr): 0.38 | | |
| Soil Temperature (°C): 7.5 | Nitrification rate: | |
| Wet & dry Acid deposition (kEq/ha/yr): | <input type="radio"/> high | |
| SO ₄ : 1.24 | <input checked="" type="radio"/> medium | |
| Cl: 2.46 | <input type="radio"/> low | |
| NO ₃ : .34 | <input type="radio"/> none | |
| Wet & dry Cation deposition (kEq/ha/yr): | | |
| NH ₄ : .35 | | |
| Ca: .31 | | |
| Mg: .47 | | |
| K: .11 | | |
| Na: 2.13 | | |

Middle Screenshot: Profile 4.1 - Vegetation Input

| | | |
|--|---------------------------------|-------------|
| Profile 4.1 | Go To Card | Check Input |
| Vegetation Input | | |
| Net uptake (kEq/ha/yr): | Canopy Exchange (kEq/ha/yr): | |
| Ca: 15 | Ca: 0.3 | |
| Mg: 12 | Mg: 0.15 | |
| K: 2 | K: 0.4 | |
| Nitrogen: 3 | Nitrogen: 0.3 | |
| <input type="checkbox"/> Uncoupled uptake | Litter Fall (kEq/ha/yr): | |
| <input checked="" type="checkbox"/> Zero Circulation (Default Circulation) | Ca: 0.2 | |
| <input type="checkbox"/> Unspecific | Mg: 0.2 | |
| Calc/Mg/K response: 2A1 | K: 0.4 | |
| <input type="checkbox"/> Include pH in response | Nitrogen: 0.9 | |
| Response Coefficient: 0.33 | Net Mineralization (kEq/ha/yr): | |
| <input type="button" value="Draw Unspecific Response Curve"/> | Ca: 0 | |
| | Mg: 0 | |
| | K: 0 | |
| | Nitrogen: 0 | |
| | Sulphur: 0 | |

Bottom Screenshot: Profile 4.1 - Soil Layer Input

| | | |
|--|-------------------------------|-------------|
| Profile 4.1 | Copy Data | Go To Card |
| Soil layer no: 3 | Clear values | Check Input |
| Soil layer height (m): 0.23 | Mineralogy: | |
| Soil water content (m ³ /m ³): 0.25 | K-Feldspars: 18 | |
| Soil bulk density (kg/m ³): 810 | Plagioclase: 12 15 [An 85] Ab | |
| Surface area (m ² /m ³): 1.4e6 | Albite: 8 | |
| CO ₂ pressure (atm): 20 | Hornblende: 1.5 | |
| % of precipitation entering layer: 48 | Pyroxene: 8 | |
| % of precipitation leaving layer: 42 | Epidote: 1 | |
| Cation uptake (% of max): 30 | Garnet: 0.1 | |
| Nitrogen uptake (% of max): 10 | Biotite: 0.5 | |
| DOC (mg/L): 8 | Muscovite: 0 | |
| pK gibbsite: 8.6 | Fe-Chlorite: 0.5 | |
| | Mg-Vermiculite: 5 | |
| | Apatite: 0.2 | |
| | Kaolinite: 0.8 | |
| | Calcite: 0.8 | |

Figure 7.47: Input cards for the PROFILE model, show as it appears to the user on the computer screen. The intention is that this model will be adapted for sustainability purposes and made available at no cost to any landowner.

| Mineral | Arrhenius factor | | | |
|-------------|------------------|------------------|-----------------|--------|
| | H ⁺ | H ₂ O | CO ₂ | Org |
| K-Feldspar | 3500 | 2000 | (1700) | (1200) |
| Plagioclase | 4200 | 2500 | (1700) | (1200) |
| Albite | 3800 | 2500 | 1700 | 1200 |
| Hornblende | (4300) | 3800 | (1700) | (2000) |
| Pyroxene | 2700 | 3800 | (1700) | (2000) |
| Epidote | 4350 | (3800) | (1700) | (2000) |
| Garnet | 2500 | 3500 | (1700) | (1800) |
| Biotite | 4500 | 3800 | (1700) | (2000) |
| Muscovite | 4500 | (3800) | (1700) | (2000) |
| Chlorite | 4500 | (3500) | (1700) | (1800) |
| Vermiculite | 4300 | 3800 | (1700) | (2000) |
| Apatite | 3500 | 4000 | (1700) | (2200) |
| Kaolinite | 5310 | 3580 | (1700) | (2000) |
| Calcite | 444 | 4000 | 2180 | 2200 |

Table 7.30: Observed and estimated temperature dependence factors (E_a/R) in $^{\circ}K$ used to adjust the rate coefficients to any temperature in PROFILE. The data were derived in Sverdrup (1990). Values within brackets are estimates.

The rate coefficients have been shown in Tab. 7.29. The upper part of the table show a approximate laboratory rate coefficients for the chemical weathering rate for primary minerals expressed as the flux of base cations (Ca, Mg, Na, K) related to a total rate expressed as keq/m^2s at $8^{\circ}C$, as applied in the PROFILE model. The values are the result of a re-evaluation of literature data from 1923-1995 and experiments carried out by the author during 1984-1996. Values within brackets have been approximated using Madelung site energies for key positions bonds which are attacked in the rate-limiting reactions, as well as applications of analogues and basic alumino-silicate crystal lattice structures. The bottom part of the table shows approximate laboratory rate coefficients for the chemical weathering rate for clay minerals expressed as the flux of base cations (Ca, Mg, Na, K) as keq/m^2s at $8^{\circ}C$ for the unaltered clay, as applied in the PROFILE model. The values are the result of a re-evaluation of literature data from 1923-1996 and experimental data by the authors during 1985-1990 and 1997-2000.

The total weathering rate for the whole soil is obtained by repeating the calculation additively for all minerals present in all layers. The weathering rate in the natural soil is calculated with rate coefficients taken from laboratory studies carried out at $25^{\circ}C$, and adjusted to soil temperatures using an Arrhenius relationship. The model appears to work excellently for a range of weathering rates (Fig. 4.1). It appears to be robust and accurate for at least all podsoils, sediment soils, clay soils and volcanic soils. The model has been frequently used in the European and the Far East Asian critical loads mapping programmes. The PROFILE model has subjected to validation tests in every country it has been used, and the performance found satisfactory in multiple sites located in 29 different countries on all continents.

Entering new kinetics or minerals

It was realized that the list of minerals available in the standard version of the model would sometimes need modification when the model is used in new geographical locations. For example, a site had 35% clay minerals, divided among three types of illite, three types of vermiculite and two types of smectite. The mineralogy list was changed by opening the mineral stack. Epidote, pyroxene and garnet was replaced with the reaction rate coefficients of illite and the stoichiometry of the three types. Biotite, apatite and vermiculite of the model were replaced with three types of vermiculite, muscovite and albite were replaced by two types of smectite. Kinetic

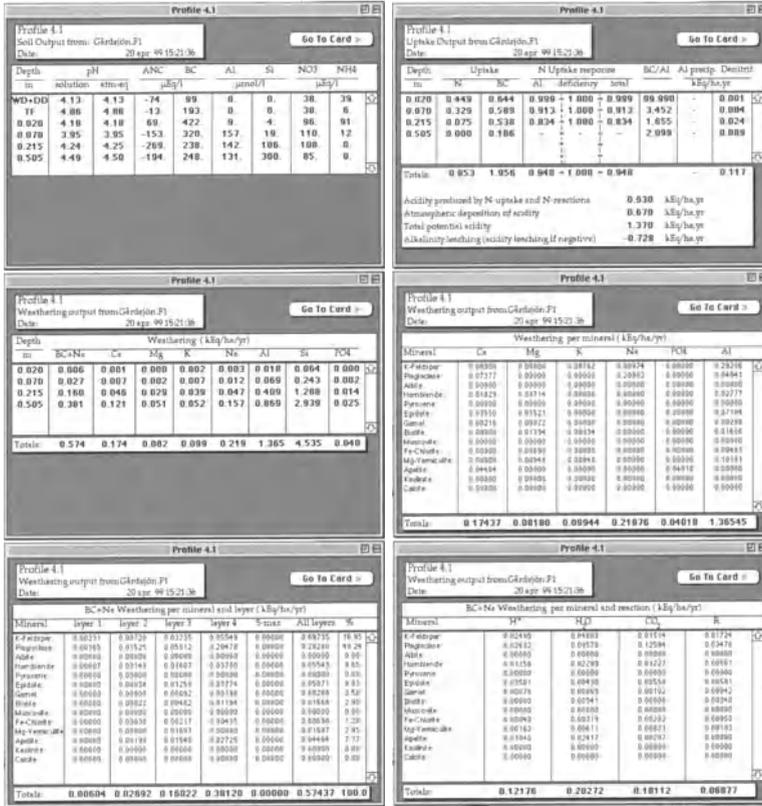


Figure 7.48: Output cards for the PROFILE model. The PROFILE model is weathering oriented in its output, something useful for sustainability assessments. From the output, the user may see the weathering rate layer-by-layer, allowing for a sustainability calculation.

coefficients and stoichiometry were adjusted. Then the stack is initiated to create another mineral kinetics input file for the executable code, and a new version of PROFILE has been created just for this site. No recompilation was necessary. The user was happy with his new PROFILE and could test out several hypotheses concerning the importance of clay minerals. The model has been modified in similar way for volcanic soils in Slovakia and Japan.

Discussion

PROFILE was used with data for the soils in catchment G1, F1, F2 and F3 in the Gårdsjön catchment, obtained from Olsson et al. (1985), the input parameters are shown in Table 7.31-7.31. The calculated rate values correlate well with the observed values, and the observed difference is within the uncertainty of the values. The weathering rate per mineral is shown in Table 7.32. It can be seen that K-feldspar, plagioclase, hornblende, epidote, vermiculite and apatite account for all the weathering in the profile. Of the individual mineral, plagioclase feldspar appears to be the most important mineral at this site. Of this the feldspars account for 60 % and hornblende and epidote account for 20 %. All mafic minerals account for approximately 30 % of the weathering rate. Clay mineral and layer silicates account for less than 10% in such

| Depth meter | Ca | Mg | K | Na | BC | P |
|----------------|---------------------------------------|-------|-------|-------|-------|-------|
| | keq ha ⁻¹ yr ⁻¹ | | | | | |
| 0-0.05 | 0.002 | 0.000 | 0.002 | 0.003 | 0.007 | 0.000 |
| 0.05-0.11 | 0.009 | 0.003 | 0.006 | 0.009 | 0.027 | 0.001 |
| 0.11-0.37 | 0.049 | 0.028 | 0.034 | 0.039 | 0.151 | 0.009 |
| 0.37-0.59 | 0.107 | 0.043 | 0.052 | 0.078 | 0.280 | 0.021 |
| 0.59-0.64 | 0.041 | 0.014 | 0.017 | 0.030 | 0.101 | 0.009 |
| Sum | 0.209 | 0.088 | 0.111 | 0.158 | 0.566 | 0.040 |

| Depth meter | Ca | Mg | K | Na | Layer sum | P |
|----------------|---------------------------------------|-------|-------|-------|-----------|-------|
| | keq ha ⁻¹ yr ⁻¹ | | | | | |
| 0-0.05 | 0.003 | 0.000 | 0.003 | 0.004 | 0.009 | 0.000 |
| 0.05-0.11 | 0.008 | 0.002 | 0.006 | 0.008 | 0.024 | 0.001 |
| 0.11-0.33 | 0.046 | 0.027 | 0.033 | 0.038 | 0.144 | 0.008 |
| 0.33-0.46 | 0.072 | 0.030 | 0.036 | 0.054 | 0.191 | 0.013 |
| Sum | 0.129 | 0.059 | 0.077 | 0.103 | 0.369 | 0.022 |

Table 7.31: *The weathering rate in keq ha⁻¹yr⁻¹ for catchment F1 (top) and G1 (bottom) calculated with PROFILE.*

soils, and only have a marginal importance for the weathering rate. The role of biotite for the present weathering rate is insignificant. Apatite account for almost 10% of the weathering rate according to the calculations, but it must be kept in mind that the kinetic coefficients for apatite dissolution are rather uncertain at present. The rate coefficients, reaction orders and product inhibition limit concentrations for the H⁺-, water-, CO₂-, and organic acid reactions for the minerals are shown in Table 7.29, the coefficients relate to the production of base cations expressed as keq m⁻²s⁻¹ at 8°C. The OH-reaction was omitted because it is insignificant in most temperate soils. The PROFILE model calculates:

- Steady state soil solution chemistry
 - pH, ANC
 - Ca, Mg, K, Na
 - NH₄, NO₃, SO₄, Cl
- Weathering
 - Weathering by mineral and layer
 - Weathering by ion and layer
 - Weathering by mineral and ion
 - Total weathering by ion for whole profile
 - Total weathering by mineral for whole profile

| Mineral | Weathering in layer, keq ha ⁻¹ yr ⁻¹ | | | | | % |
|-------------|--|-------|-------|-------|-------|--------|
| | 1 | 2 | 3 | 4 | Sum | |
| K-feldspar | 0.003 | 0.006 | 0.025 | 0.029 | 0.062 | 16.8 % |
| Plagioclase | 0.006 | 0.014 | 0.060 | 0.082 | 0.161 | 43.6 % |
| Hornblende | - | 0.001 | 0.015 | 0.018 | 0.034 | 9.2 % |
| Epidote | - | 0.001 | 0.009 | 0.028 | 0.038 | 10.3 % |
| Garnet | - | - | 0.001 | - | 0.001 | 0.3 % |
| Biotite | - | - | 0.002 | 0.002 | 0.004 | 1.1 % |
| Chlorite | - | 0.001 | 0.004 | 0.004 | 0.009 | 2.4 % |
| Vermiculite | - | 0.001 | 0.017 | 0.007 | 0.024 | 6.5 % |
| Apatite | - | - | 0.013 | 0.023 | 0.035 | 9.5 % |
| Sum | 0.009 | 0.024 | 0.144 | 0.191 | 0.369 | 100 % |

Table 7.32: *The weathering rate for each mineral in the soil in keq ha⁻¹yr⁻¹ for catchment G1, as calculated with PROFILE.*

– Total weathering by reaction (hydrogen ion, hydrolysis, organic acids, carbon dioxide) by mineral for whole profile

- Acidity effect on roots
- Any shortage of base cations for nutrient uptake

The effect of different epidote soil contents has been investigated with the model using the Gårdsjön catchment F1 as example. The basic input data can be found on the PROFILE input cards shown in Fig. 7.48. The results from increasing the epidote soil content from 0 to 100% cause the weathering rate to increase from 0.52 keq ha⁻¹yr⁻¹ when the epidote content is 0%, through 0.57 keq ha⁻¹yr⁻¹ at the actual 1%, at 5% 0.67 keq ha⁻¹yr⁻¹, at 25% it is 1.70 keq ha⁻¹yr⁻¹ to 8.80 keq ha⁻¹yr⁻¹ when the epidote content is 100%. The model performance with respect to epidote can actually be tested on field data. In Maryland, a number of sites are available showing a large span in epidote content. The results of applying the model to these sites is shown in Tab. 7.31 and 7.32. The representation of epidote in the model do appear to be adequate for predicting field rates with reasonable success as compared to the field observations. The weathering rate in nature is determined by the abundance of weatherable minerals and their texture in the soil. The PROFILE model represents the first tool available for estimation of the weathering rate from independent data on geochemistry and soil conditions. Accordingly, an understanding of the soil mineralogy and texture distribution on a regional scale is essential for understanding the conditions for weathering available in the region.

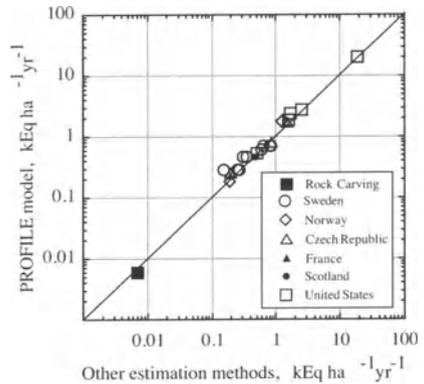


Figure 7.49: *The PROFILE model has been successfully tested at a number of sites inside and outside Sweden.*

4.1

| F1 | F2 | F3 | G1 | Method |
|---------------------------------------|------|-----------|------|---|
| keq ha ⁻¹ yr ⁻¹ | | | | |
| 0.54 | 0.61 | 0.59 | 0.36 | Budget study |
| 0.57 | 0.60 | 0.67 | 0.37 | Laboratory kinetics, PROFILE |
| 0.53 | 0.60 | 0.61 | 0.38 | Total analysis correlation |
| 0.59 | - | - | 0.39 | Sr isotope ratio method |
| 0.56 | 0.60 | 0.64 | 0.38 | Average present value |
| 0.74-0.81 | - | 0.81-1.31 | 0.35 | Historic rate, bedrock mineralogy reference |
| 0.53 | - | - | 0.30 | Historic rate, C-layer mineralogy reference |
| 0.74 | - | - 1.08 | 0.30 | Historic rate, laboratory kinetics, SAFE |
| 0.47 | 0.47 | - 0.48 | 0.37 | Historic conditions, present soil, PROFILE |

Table 7.33: Summary of base cation release rates due to chemical weathering in keq ha⁻¹yr⁻¹ at different catchments at Lake Gårdsjön, Sweden, using different methods. It can be seen that the different methods give approximately the same results.

7.8.2 Calculating field weathering rates

Weathering can be estimated for field conditions by two existing models.

- The "Olsson-Melkerud" model (Olsson and Melkerud, 1990; Sverdrup et al., 1990), relates total analysis to historical weathering rates for podsols.
- PROFILE, developed at the University of Lund in Sweden (Warfvinge and Sverdrup, 1990; Sverdrup and Warfvinge, 1992). The model is new in the sense that it is the first model successfully to calculate field weathering rates from geochemical properties of the soil such as texture and mineralogy, and that the input data all can be measured on soil samples.

The above mentioned models are mutually consistent, they are the only models to survive a test against field data, and have been found to operate with reasonable accuracy. PROFILE is a steady state model designed originally to solve soil acidification problems. This implies that the whole path from original pre-acidification state to the post-acidification state is circumvented, and the final state calculated directly. At steady state, all sources of input acidity to the system are balanced by either internal sources such as weathering, and nitrate uptake or export of acidity from the system. The definition of steady state implies that the net depletion of the ion exchange complex for alkalinity is zero. The soil processes affecting major constituents and the alkalinity-acidity balance in the soil are many.

7.8.3 Estimating soil mineralogy inputs to the model

The model UPPSALA is a normative back-calculation model for reconstructing the mineralogy from the total analysis in order to provide input to models like PROFILE from simple survey data. The method find a soil mineralogy which is consistent with the observed total analysis by simple mass balance, assuming standard stoichiometries for the soil minerals. For the models to be applicable at single forest properties, mineralogy must be obtainable using data that can be collected at low cost. We have developed such a methodology and it has been successfully tested at the regional level using data from the Swedish forest inventory.

Input data to the UPPSALA model for converting total analysis to mineralogy is Na_2O , CaO , MgO , K_2O , P_2O_5 , Al_2O_3 , SiO_2 and Fe_2O_3 as % weight. Total analysis is often available from national forest inventories, many Geological Surveys have also collected such data across whole regions or nations. Agricultural soil surveys may also be of help. The optimal strategy is to run the total analysis first, and then selecting the sampling sites for mineralogy. The UPPSALA model is based on a priori knowledge of the stoichiometric composition of the minerals of the particular soil. The minerals have been grouped into assemblies of minerals with similar composition and dissolution rate. It is assumed that the following mineralogical groupings are valid; Muscovite is assumed to comprise muscovite, secondary di-octahedral illite, di-octahedral chlorite and vermiculite of secondary weathered type. Chlorite comprises tri-octahedral chlorite, primary illite and tri-octahedral vermiculite of primary type as well as biotite and phlogopite and glauconite. Hornblende implies all amphiboles, such as hornblende, riebeckite, arfvedsonite, glauconite and tremolite. Epidote comprises all epidotes, zoisites and pyroxenes. Plagioclase has been assumed to be oligoclase with 80% albite feldspar component. K-feldspar is assumed to be 10% albite feldspar component. All phosphorus has been attributed to apatite. The equations used in the UPPSALA model for calculation of the % weight content of the individual minerals are given below. The UPPSALA model is a general model for whole Sweden, whereas the BERNE variant of the UPPSALA model is a regional specialization and a generalization in each region, where individual models have been developed for each one of 15 normative provinces, those are shown in Fig. 7.51. Later analysis have increased this to three provinces. In the simple UPPSALA model the following calculation sequence is performed:

$$\begin{aligned}
 \text{K-Feldspar} &= \max(0, 5.88 \cdot \text{K}_2\text{O} - 0.588 \cdot \text{Na}_2\text{O}) \\
 \text{Plagioclase} &= \max(0, 11.1 \cdot \text{Na}_2\text{O} - 0.22 \cdot \text{K-Feldspar}) \\
 \text{Apatite} &= 2.24 \cdot \text{P}_2\text{O}_5 \\
 \text{Hornblende} &= \max(0, 6.67 \cdot \text{CaO} - 3.67 \cdot \text{Apatite} - 0.2 \cdot \text{Plagioclase}) \\
 \text{Muscovite} &= \max(0, 2.08 \cdot \text{K}_2\text{O} - 0.208 \cdot \text{Na}_2\text{O}) \\
 \text{Chlorite} &= \max(0, 3.85 \cdot \text{MgO} - 0.39 \cdot \text{Hornblende} - 0.39 \cdot \text{Muscovite}) \\
 \text{Epidote} &= \max(0, 0.1 \cdot \text{Hornblende} + 0.03 \cdot \text{Plagioclase} - 0.3) \\
 \text{Quartz} &= \text{SiO}_2 - 0.63 \cdot \text{Plagioclase} - 0.68 \cdot \text{K-Feldspar} - 0.38 \cdot \text{Muscovite} - 0.33 \cdot \text{Chlorite} - 0.45 \cdot \text{Hornblende} - 0.42 \cdot \text{Epidote} \\
 \text{Al-residual} &= \text{Al}_2\text{O}_3 - 0.1 \cdot \text{Plagioclase} - 0.1 \cdot \text{K-Feldspar} - 0.26 \cdot \text{Muscovite} - 0.09 \cdot \text{Chlorite} - 0.01 \cdot \text{Hornblende} - 0.025 \cdot \text{Epidote} \\
 \text{Delta} &= \text{Quartz} + \text{Plagioclase} + \text{K-Feldspar} + \text{Muscovite} + \text{Chlorite} + \text{Hornblende} + \text{Epidote} + \text{Apatite} \\
 \text{Vermiculite} &= \min(\text{Al-residual}, \max(0, (100\% - \text{Delta})))
 \end{aligned}$$

For each of the geological province the observed mineralogy was correlated to total analysis. Calcareous sites must be separated, and after K-feldspar, plagioclase and apatite have been calculated, the following is applied: The calculation is checked by calculating the amount of

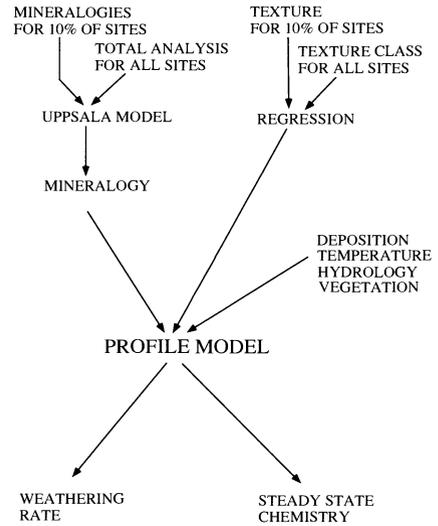


Figure 7.50: The sequence for calculating field weathering rates using a regionally distributed sampling of soil oxides and a subset of measured mineralogies.

quartz, and only such sites which lie within the range 95-105% are accepted. The following equation is used.

$$\text{Calcite} = \max(0, 1.79 * \text{CaO} - 3.67 * \text{Apatite} - 0.2 * \text{Plagioclase})$$

The model was calibrated with a small number of measured mineralogies for each province. This calibration has limitations, as the accuracy of field mineralogy is limited and not necessarily better than the normative reconstruction. The BERNE version of the UPPSALA model has also been successfully applied to Switzerland, an area with great geological complexity. A fully codified version is envisioned in the second phase of the SUFOR Programme. Normally, the first UPPSALA version of the model calculation can be carried out in a simple spreadsheet or for a single site, with a pocket calculator. When no more adequate ions available for the formation of a mineral, then the content of that mineral is set to zero. The UPPSALA model has so far been tested and validated in Sweden (granitic), Switzerland (mixed geology from glaciated granitic to non-glaciated and sedimentary rocks) and Maryland (sedimentary rocks and fluvial marine deposits). Fig. 7.53 show the correlation between calculated and observed mineralogy. For K-feldspar we have $r^2=0.76$, plagioclase $r^2=0.69$, hornblende $r^2=0.41$, epidote $r^2=0.41$, chlorite $r^2=0.52$, pyroxene $r^2=0.5$. The regression for Apatite is equal in all provinces and should be corrected from the above to:

$$\text{Apatite} = 11.5 \cdot P_2O_5 - 0.92 \quad (7.169)$$

The implication is that the mineral phase containing phosphorus contains less phosphorus than pure apatite. Then the correlation coefficient increase to $r^2=0.85$. The results of the Berne model are significantly better than the results of the simpler Uppsala model. These differences do not come through that strongly to weathering, much because some of the variables and errors are dependent and cancel in the weathering rate calculation. The team in Sweden and our colleagues are the only ones that can perform this kind of calculations. Our results show that there are two types of plagioclase in Swedish soils. One with 90% albite and another more frequently occurring with approximately 85% albite. n is the fraction anorthite in the plagioclase solid solution. A matrix of 9 different oxides determines the mineralogy of 9 independent minerals: depending on the vector of minerals \vec{M} being equal to the vector of oxides \vec{O} times the eigenmatrix M_E of the mineralogy-oxide equation system:

$$\vec{M} = \vec{O} \cdot E_M \quad (7.170)$$

There are some constraints; only hornblende, pyroxene and apatite are assumed to contain phosphorus, only plagioclase is assumed to contain sodium (Na) and magnesium is excluded from K-feldspar and plagioclase. Finally, apatite is considered to consist of phosphorus and calcium only. Finally, a dependent sequence is performed to estimate quartz and undefinable precipitates, residues and clays (Al-residual), everything is finally called "Vermiculite".

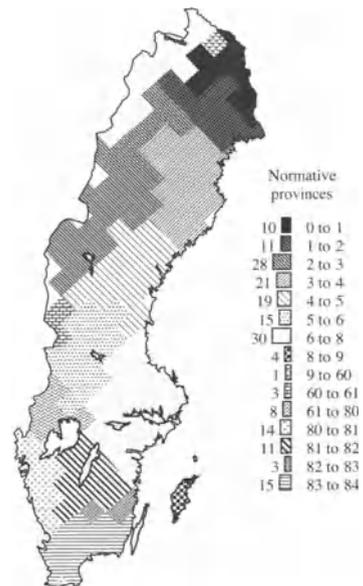


Figure 7.52: The UPPSALA model initially assumed that Sweden could be modelled as one geological province.

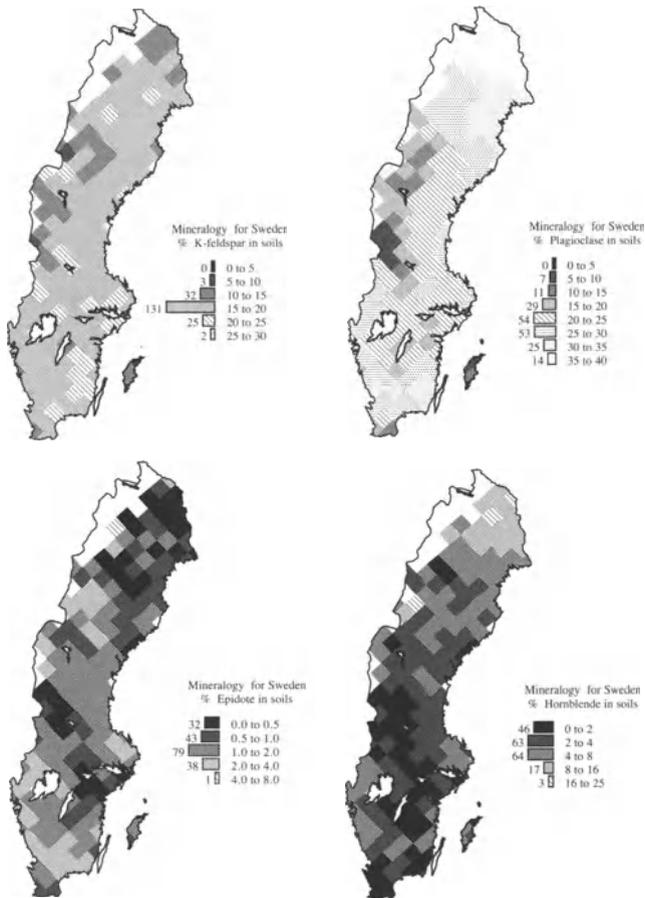


Figure 7.51: The calculated forest soil content of K-feldspar, plagioclase, epidote and hornblende as percent weight using the BERN model, a variant of the UPPSALA model.

7.8.4 Including clay minerals in the model

From transient evolution to steady-state

Our results from dissolution experiments on primary minerals and clay minerals (Sverdrup 1990; Holmqvist 2001) confirm that the development of a mineral surface with a basic aluminosilicate structure goes through different steps before it reach a steady-state condition, and that these steps can take such a long time that evolution under natural conditions cannot always be neglected. This hypothesis was also put forward by Rimstidt (1997; 1999). These steps involve evolution of different layers with different ratio of base cations/alumino-silicate at the mineral surface. The thickness and maturity of these layers develop according to the environment. In highly acid environment (solution) are these layers more depleted in base cations and also thicker than in more natural environment. The same phenomena maybe true for environment which also are increasing the dissolution rate, high concentration of OH, CO₂, organic ligands. These depleted layers have a significant effect on the base cation release when the chemical

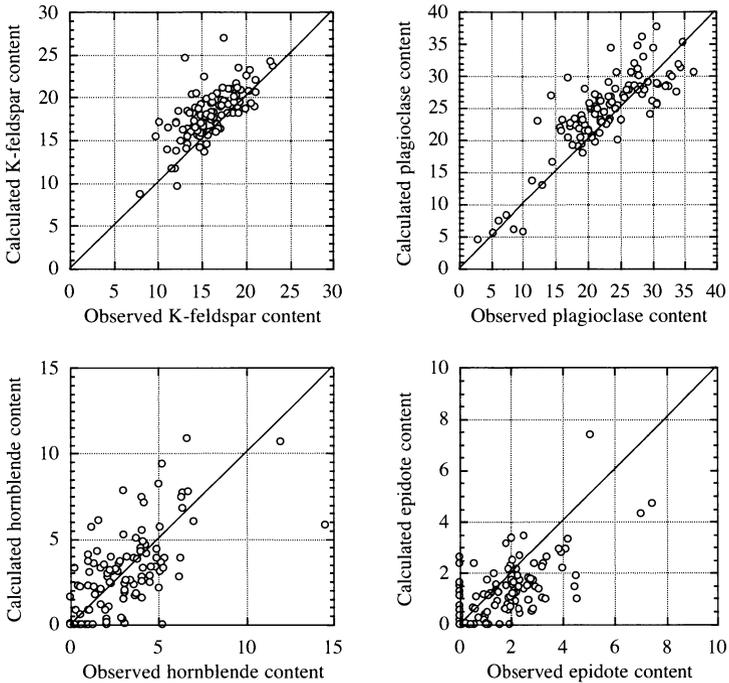


Figure 7.53: *Test of the Bern model. The correlation between observed and estimated is for quartz $r^2=0.85$, apatite $r^2=0.85$, K-feldspar $r^2=0.76$, plagioclase, $r^2=0.69$, epidote $r^2=0.41$ and hornblende $r^2=0.41$.*

environment is changing (H^+ , OH^- , CO_2 , organic ligands-concentration). If the surrounding environment change from less acidic to more acidic the adjustment is relatively fast. The reason for this are that the hydrogen ions, now in higher concentration, will attack the available base cation sites which haven't been exposed before in such extension. But, if the condition are moving from a acid to less acidic environment it will take considerable time before it reach its steady-state condition. According to our hypothesis this is an expected event. The highly acidic solution have develop relatively thick depleted layers and when the environment are changing to a less environment it will take considerable time before the depleted layer have been dissolved, and reached a state when the new less hydrogen concentration can effect/reach the unaffected mineral structure. Not until the former "thick" depleted layers have been recovered to the new steady-state according to the new environment/circumstances.

The weathering mechanisms described above can have important implications on the dynamic soil chemistry. The soil solution will only seldom reach a true steady state, but ever change towards new final states as the basic conditions change through the seasons, years and decades. The soil solution composition in this respect reflect the outer environment and transfers it to the reaction at the mineral surface. The soil solution in its turn depends on several conditions such as climate, vegetation, wet and dry deposition. Therefor can this approach on chemical dissolution of aluminosilicate minerals have significant impact how soil solution react on natural and/or anthropogenic causes. During dissolution, a steady state situation builds up at the surface. At steady state we get an outer layer, depleted in base cations, and intermediate

| Mineral | Interlayer | octahedral | tetrahedral |
|-----------------|---|--|---|
| Muscovite | K | Al ₂ | AlSi ₃ O ₁₀ (OH) ₂ |
| illite 1 | K _{0.7} Mg _{0.05} | Al | AlSi ₃ O ₁₀ (OH) ₂ |
| illite 2 | K _{0.6} Mg _{0.05} | Al | AlSi ₃ O ₁₀ (OH) ₂ |
| illite 3 | K _{0.5} Mg _{0.05} | Al | AlSi ₃ O ₁₀ (OH) ₂ |
| Kaolinite | | | |
| Chlorite | Ca _{0.5} Mg _{1.5} | Mg _{1.5} Fe _{0.5} Al | AlSi ₃ O ₁₀ (OH) ₈ |
| Montmorillonite | Ca _{0.25} | Al _{1.5} Mg _{0.5} | Si ₄ O ₁₀ (OH) ₂ |
| Smectite 1 | Ca _{0.075} Mg _{0.03} K _{0.03} | Al _{0.01} | AlSi ₃ O ₁₀ (OH) ₂ |
| Smectite 2 | Ca _{0.03} Mg _{0.02} K _{0.01} | | AlSi ₃ O ₁₀ (OH) ₂ |
| Kaolinite | | | |
| Biotite | KMg ₂ | Al _{0.5} Fe _{0.5} Mg | AlSi ₃ O ₁₀ (OH) ₂ |
| Vermiculite 1 | Ca _{0.075} Mg _{0.075} K _{0.15} | Al _{0.05} | AlSi ₃ O ₁₀ (OH) ₂ |
| Vermiculite 2 | Ca _{0.05} Mg _{0.063} K _{0.13} | Al _{0.05} | AlSi ₃ O ₁₀ (OH) ₂ |
| Vermiculite 3 | Ca _{0.02} Mg _{0.05} K _{0.1} | Al _{0.05} | AlSi ₃ O ₁₀ (OH) ₂ |
| Kaolinite | | | |
| Feldspar | (K,Na,Ca) | | AlSi ₃ O ₈ |
| Sericite | Na _{0.1} K _{0.75} | Al _{1.9} Mg _{0.1} | Al _{0.84} Si _{3.16} O ₁₀ (OH) ₂ |
| Illite-verm | Ca _{0.05} Mg _{0.02} K _{0.3} | Al _{0.02} | AlSi ₃ O ₁₀ (OH) ₂ |
| Vermiculite 2 | Ca _{0.02} Mg _{0.05} K _{0.1} | Al _{0.05} | AlSi ₃ O ₁₀ (OH) ₂ |
| Kaolinite | | | |

Table 7.34: *Alteration sequence used in the model based on actual mineral stoichiometries found at the German Adenau site. The sequence starts with a primary mineral; feldspar, muscovite, biotite, chlorite and ends up with secondary alteration phases such as illite, vermiculite or smectite, where the endpoint is kaolinite or amorphous residuals, completely depleted in base cations. It is possible that the sequence starting with feldspar could be simplified to go straight to smectite and kaolinite. This sequence could probably also be allowed to start with amphiboles or epidotes. The most depleted minerals are those exposed to weathering for the longest time, these are found most depleted in the top of the soil and depletion increase with soil age.*

layer partially leached and the inner unchanged mineral core. At steady state the dissolution of the outer layer carefully balances the rate of leaching of the unchanged core, and the thickness of these layers are controlled by the outer chemical conditions. Accordingly, when these change, then the layers must also adjust. When moving from acid solution to less acid, the completely leached layer must decrease and the partly leached layer expand into the leached layer. These adjustments can only occur very slowly, and may for periods of time decrease net base cation release most significantly. In layered minerals, the situation is conceptually confounded by the fact that this behaviour may be distributed onto alternating layers with fundamentally different properties. We should further imagine that this may occur in asymmetric minerals, where a number of these situations are stacked in layers on top of each others.

Alteration sequences for micas and feldspars

The indications for alteration of micas to secondary clays are many; (i) experimental studies in which micas have been transformed to expansible 2:1 minerals through the extraction of

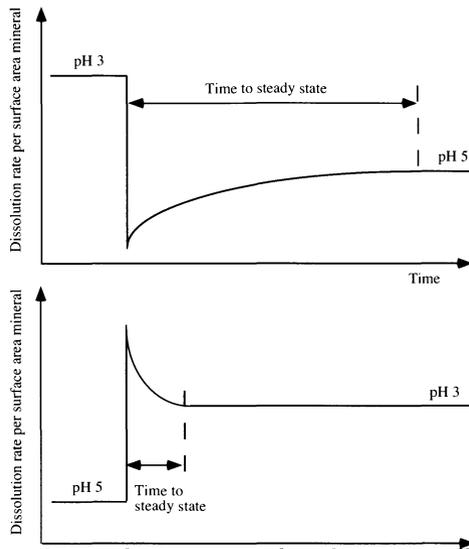


Figure 7.54: Moving from less acidic to more acidic solutions, usually adjusts to the new conditions relatively fast. The adjustment when the conditions move from acidic to less acidic conditions differ very much from the scenario described above and take considerable longer time before it have come to a new steady state. The time involved in the adjustment after a change from less acid to more acid is in the order of days and weeks, whereas the adjustment time for a change from acid to less acid may range from several weeks to years. If we use dynamic models to simulate soil chemistry during a growth season, water chemistry within the year or in time periods of months to years, this could be important. How important remains to be determined.

interlayer potassium with various chemical and biological agents, and (ii) weathering studies of clay minerals in soils developed under natural condition that show transformation of micas have taken place as a function of depth, time, rainfall, and (iii) studies of mineralogy changes with time and with soil depth in the profile (Dixon and Weed, 1989). It has been concluded that the influence of pH, salt concentration, kinds and activity of various ions in the soil solution, layer charge and time, are among the essential factors (Carstea, 1968; Churchman, 1980). These influences of the parent material, however, are only effective in co-operation with environmental factors such as temperature, precipitation and wetting and drying cycles (Churchman, 1980; Rich and Cook, 1963). It seems that the weathering of dioctahedral micas is greatest when the parent material is interstratified with vermiculite. It is suggested that this is in part a continuation of a more rapid weathering rate already naturally exhibited by the parent mica as it weathered from the fresh schist to the soil parent material of the C horizon (Rich and Cook, 1963).

Transformation of K-bearing expandible 2:1 minerals are due to alteration of interlayered cations, predominantly potassium, by hydrated cations such as calcium and magnesium. The important mechanisms to this transformation of mica are; (i) diffusion of K^+ out and diffusion of the counter ion to the vacated spot (Chute and Quirk, 1967; Reed and Scott, 1962), (ii) oxidation of ferrous iron in the octahedral position lowers the layer charge (Farmer et al., 1971; Taylor et al., 1968; Weaver and Pillard, 1973) and (iii) hydroxyl orientation, Norrish 1972). Wilson and Nadeau (1984) studies on interstratified clay minerals suggest from TEM analysis that the thin illite and chlorite crystals found in diagenetic clays have precipitated directly from solution and must be regarded as neoformed. Trioctahedral vermiculite is an alteration product of biotite, phlogopite or chlorite and accumulation of this vermiculite are optimized under mod-

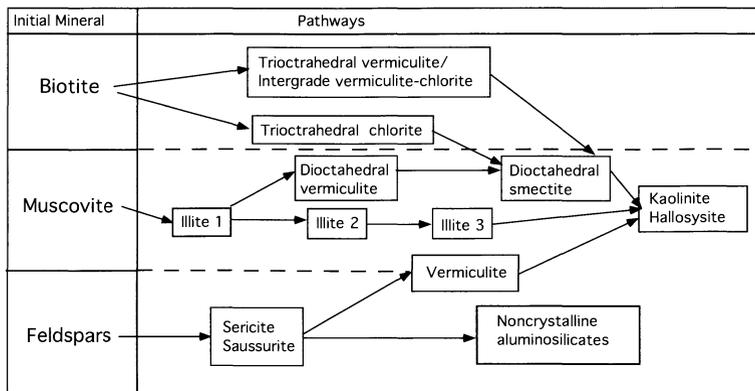


Figure 7.55: A simplified summary over the weathering pathways for the micas and the feldspars. We suggest that there are at least three major pathways determined by whether the starting point is 1) a trioctahedral biotite mica, 2) a diocahedral muscovite type of mica or 3) a feldspar. The literature is not internally consistent with the definition of sericite, but we define it as the first alteration product of feldspar weathering.

erate weathering condition. Diocahedral vermiculite is mainly formed from muscovite and are more frequent in environment of more severe weathering condition. Under intensive weathering environments, acid soil conditions, vermiculite invariably have a hydroxy-Al-interlayer (Rich, 1958; Holmqvist, 1994). Vermiculite is maybe the most common clay mineral that will form in Scandinavian soils where either biotite or chlorite is present in the parent material. For vermiculite to persist under intense weathering conditions or for long periods of time, the normal weathering sequence is:

Biotite \longrightarrow vermiculite \longrightarrow hydroxy-al-interlayered vermiculite

It seems likely that eventually the biotite-derived vermiculites either dissolve to furnish essentially the same ions in solution that the parent mica, or they may alter internally toward some mineral that is stable under the prevailing solution condition, such as montmorillonite. The pathway of formation of pedogenic chlorite from biotite is uncertain. Suggested pathways include direct formation from biotite and an indirect origin via vermiculite. According to Churchman (1980) weathering of chlorite appeared to follow the sequence: chlorite \longrightarrow interlayered hydrous mica \longrightarrow chlorite-swelling chlorite \longrightarrow chlorite-vermiculite, with further weathering leading to decomposition of chlorite layers. The extent of alteration of biotite and chlorite increased with increase in precipitation, beech forest vegetation, and time, and decreased with decrease in temperature and tussock grassland vegetation (Churchman, 1980). Influx of mineral colloids and solutes of Si, Fe, Al, Mg, Ca, and Na (with limited K) into swamps and basins frequently has resulted in the formation of smectite during the Quaternary.

Proposed mechanisms for dissolution of clay minerals

The most common minerals of Northern European that can be found as precursor for the most abundant pedogenic clay minerals in soils can be divided into four types, and hence four basic mineral lattice destruction rates, based on the (1) muscovite lattice going to illite, (2) biotite going to vermiculite/chlorite, (3) feldspars going to sericite and vermiculite (Holmqvist, 1994) and (4) chlorite going to montmorillonite and smectite. The base cation release rate is determined by the mineral lattice destruction rate and the stoichiometric content of base cations

| Layers | 1 | 2 | 3 | 4 | 5 |
|--------------------|---|------|------|------|------|
| Mineral | | | | | |
| K-Feldspar | 2 | 3.5 | 3.5 | 5.5 | 5.5 |
| Plagioclase | 2 | 0.5 | 0.5 | 0.5 | 0.5 |
| Hornblende | 0 | 0.01 | 0.03 | 0.05 | 0.05 |
| Smectite | 0 | 5.5 | 0 | 0 | 0 |
| Muscovite | 0 | 0 | 0 | 0 | 0 |
| Illite-1 | 0 | 0 | 0 | 0 | 38 |
| Illite-2 | 0 | 0 | 1.5 | 28 | 0 |
| Illite-3 | 0 | 24 | 27.5 | 0 | 0 |
| Chlorite | 0 | 0 | 0 | 0.5 | 2 |
| Illite-vermiculite | 0 | 2 | 1.5 | 0.5 | 0 |
| Biotite | 0 | 0 | 0 | 0 | 0 |
| Vermiculite-1 | 0 | 0 | 0 | 7 | 2 |
| Vermiculite-2 | 0 | 0 | 0.5 | 8 | 0 |
| Vermiculite-3 | 0 | 4 | 13 | 0 | 0 |

Table 7.35: *Alteration sequence found at the Adenau site. It can be seen that the most depleted alteration forms are found in the top of the soil. It is also evident how feldspars alterates to a dioctahedral smectite, muscovite through a series of illites, how the interlayered chlorite weathers to a mixed clay and how biotite evidently produce a series of vermiculites. This kind of datasets form a basis for forming our hypothesis.*

in the lattice. The kinetic coefficients were determined by working with the alteration series and empirical clay stoichiometries were used. The base cation release from clay minerals in soils are assumed to be controlled mainly by two factors. First it is assumed that the weathering kinetics is determined by the basic aluminosilicate structure of the clay mineral. Secondly that the base cation release rate is determined by the amount of base cation released per unit basic lattice destroyed per unit time.

$$BC \text{ release rate} = (\text{Lattice destruction rate}) * (\text{Mineral BC content}) \quad (7.171)$$

The "lattice destruction rate" we define as the long term decomposition rate of the most resistant lattice structure and "stoichiometry" we define as the elemental composition of the lattice. The lattice structure will be the same for many clays and based on the top member of the clay genesis series. Thus we will have a series of illites starting with muscovite.

The SAFE and PROFILE models are available at [HTTP://WWW2.chemeng.lth.se](http://WWW2.chemeng.lth.se) for both Apple and PC computers, free of cost.

7.8.5 Some persistent myths in weathering

Weathering is a field that has been affiliated with myths and persistent disinformation, and well as a large amount of experimental research of such quality that interpretation was impossible. Much is caused by the lack of systems approach and the development of proper models before

experiments are started, often supported by proponents of the myths of weathering. We think it is time to take a critical look at some of these myths, and make clear statements about them. We know that this will offend and insult many, but we think this kind of inspection is very necessary, as well as a matter of honesty. These hypotheses have been around for some time, and it is only fair that they should be critically reviewed with respect to other theoretical and experimental developments. Since the beginning of this century, the kinetics of weathering has been researched and but for decades most comparisons of different results indicated that laboratory results were inconsistent with field observations (Paces, 1983; Velbel 1986). For many years, weathering research made little or no progress, much caused by lack of theoretical analysis, but also a paradigm of describing weathering in terms of reversible equilibrium between water solutions and silicate minerals. Many of the old laboratory results were internally inconsistent, but this went undetected for a long time and thus the prospects for progress looked dismal. Despite impressively large and increasing voluminous articles during the 1970's and the 1980's on the theory of chemical weathering, the different proposed models were totally unable to calculate reasonable field weathering rates (Velbel 1986; Sverdrup 1990). The American author Mark Twain once made the satirical remark;

"So many Honorable Investigators have shed so much Darkness upon this Subject, that We shall soon know Nothing about it."

which would fit the past and partly present situation of weathering research well. The situation is slowly improving, but still many myths, prestige and conservatism is abundant. As a result, many geochemical researchers have become notoriously suspicious of mathematical models for natural systems, often for very good reasons considering the history. This was especially true for the earlier weathering models, which in the past have a reputation of total failure to produce what nature does. After 1985, new scientific advances improved the situation a lot, acidification research contributed much to a new momentum forward (Sverdrup and Warfvinge 1988, Sverdrup 1990). In weathering, unfortunately, several unsubstantiated myths have been very persistent, and because these myths show up again and again, we need to deal with them directly. For all the hypotheses inspected here, we may give many references, but we will not do this as the purpose is not to placate colleagues that may have made mistakes, but rather to inspect ideas and reject those that do not hold up.

Organic acids ?

The first myth is that "Organic acids are very important for the field weathering rate". Time and again, this myth has been propagated as a reason for doing more research; organic acids were claimed to be the only reactant that mattered. It has survived mainly due to conservatism in experimental designs, and an inability to analyze results with the models required for multi-reaction, multi-phase chemical systems. Famous researchers in weathering have failed completely to realize that several reactions were acting simultaneously, rendering experimental designs hopelessly inadequate. Experimental data deliver a clear and loud message; Quantitatively, the reactions with H^+ and water dominate. Organic acids have their greatest influence on the weathering rate through the complexing of Al and solution equilibria (Sverdrup, 1990).

Laboratory rates don't work out there ?

The second myth is that "Laboratory rates of weathering cannot be reconciled with field rates since laboratory rates are several orders of magnitude larger". Again this myth persists because many researchers in geochemistry treat weathering as the only process existing in the world. Let's be clear about it; this is definitely not the case. Many also persist in ignoring the difference in reaction conditions between field experiments and field conditions when they make their comparisons. This is a mistake on a very elementary level, and not at all acceptable in scientific research. If the difference in chemical conditions, temperature, hydrology and

interaction with other soil processes are considered carefully, then laboratory rate coefficients can be used to calculate field rates with high accuracy (Sverdrup and Warfvinge, 1988; 1990; 1993, Sverdrup 1990). Laboratory weathering rates are consistent with field weathering rates, if your results show something else, you probably made a mistake.

Is mycorrhiza growing at the mountains roots ?

The third myth "Weathering under field conditions in forest soils is driven by direct attack by mycorrhiza". This myth is persisting because it has become an efficient research funding vehicle, despite its obvious lack of support in scientifically sound data, as well as its blatantly overlooking of large amounts of published data relevant to the subject. Against this myth, hard thermodynamic evidence is available, and to cite the famous theoretical physicist Sir Arthur Eddington; "If a theory have been found to be against the second law of thermodynamics, then there is absolutely no hope for it, but only certain humiliation". Many persons have placed their faith in the concept of mycorrhiza weathering, and they will not be happy to hear what is to follow here.

We will jump right to the critical issue: Does fungi selectively dissolve stone ? How much of the weathering we can observe in forested catchments is really connected to the action of mycorrhiza ? The mycorrhiza hypothesis has been forwarded in several articles, and also in the well known science magazine Nature, so we need to take a closer look before we can safely lay it to rest. The hypotheses forwarded are:

1. Mycorrhiza hyphae penetrate soil mineral particles and dissolve minerals by organic acid exudation. It is hypothesized for weathering under field conditions that a substantial part of the total rate occur this way.
2. The dissolution is characterized by the reaction with organic acids. The hypothesis states that the weathering rate is strongly enhanced by exudation of organic acids like oxalic and citric acids.
3. The dissolution rate will be independent of mineral surface area. The mycorrhiza hyphae will encounter mineral grains at random and penetrate them.
4. Mycorrhiza hyphae penetrate soil mineral particles and dissolve minerals by organic acid exudation. It is hypothesized for weathering under field conditions that a substantial part of the total rate occur this way.

The temperature dependency factors in the Arrhenius equation have been accurately determined in laboratory studies for different reactions, and they are:

- Reaction with hydrogen ions: 3800-4200°K
- Reaction with water molecules: 2500-3800°K
- Reaction with organic acids: 1200-2000°K
- Reaction with carbon dioxide: 1700-2200°K

The temperature dependence has been tested in the field: (1) Temperature transect on uniform mineralogy in the Swiss Alps on a forested mountain slope by Drever and Furrer (1993) gave the result 3600-3400°K. (2) A study of hornblende weathering rates from Maryland to Baffin land by Sverdrup (1990) gave the result 3800°K. We may conclude that the dissolution rate under field conditions is dominated by other reactions than the reaction with organic acids. The dominating reaction under field conditions appear to be the reaction with hydrogen ions and the water reaction. The reaction with organic acids appear to play a minor role. In their article in Nature van Breemen et al., (1998), report that something like 150 meter of

holes in soil minerals with a diameter of 0.3 to 10 micrometers are produced per year in the soil. Further this is hypothesized to take place in the O and E-horizons of the soil. This corresponds to a total base cation production of 0.21 kg base cations per ha yr for an average hole diameter of 20 microns to 0.002 kg of total base cation flux per ha and yr for an average hole size of 0.6 micron. It is well established that only a small part of soil profile weathering takes place in this part of the soil, and whatever effect mycorrhiza has here, the total effect on the whole rate will be minuscule. For organic acids to increase the rate over such a small proportion of the total mineral surface in the soil, the organic acid concentration would have to be increased to 3,000-10,000 times the concentration in the soil if the soil is dominated by feldspar, something which would kill the mycorrhiza itself within seconds. If the soil is pure pyroxene or amphibole 100 to 500 times increase would be sufficient. Thus it appears that the authors of the hypothesis have been grossly exaggerated the effects seen and have failed to test the reasonability of their interpretation. Sverdrup et al, (1992), studied the weathering rate along a transect on uniform mineralogy in Maryland on soils covered by deciduous forest, going from a texture of 6 million m²/m³ soil to 0.5 million m²/m³ soil. In this interval, weathering rate was linearly dependent on soil mineral surface area. Velbel (1987) investigated dependence of weathering rate on soil texture, using North American studies and found a linear relationship. Sverdrup (1990) investigated dependence on soil texture and exposed mineral surface, using results from Swedish weathering studies. Weathering rates under field conditions appear to be linearly dependent on the mineral surface area; the soil texture. The hypothesis that mycorrhiza is quantitatively important for weathering can be firmly falsified;

1. The weathering rate is not independent of soil texture or total mineral surfaces.
2. No amount attributable to fungal action or mycorrhiza interference have ever been found in a budget study. In fact the margins for adding in extra weathering in such budgets are extremely small.
3. The temperature dependencies indicate that organic acids account for a minor part of the total weathering rate (5-15%).

Does mycorrhiza fungi dissolve stone? Yes, from the perspective of the fungus, they do, but no, from the perspective of quantitative amounts on the forest stand level. The volume of the total dissolution that is caused by this pathway do not reach significant amounts. Amounts of 0.2 kg ha⁻¹yr⁻¹ released through the action of fungi must be set against the total observed amounts recorded in field experiments and catchment budgets; From 5 kg ha⁻¹yr⁻¹ up to more than 50 kg ha⁻¹yr⁻¹. Estimation of historic weathering rates ought to include any occurring weathering caused by mycorrhiza, yet such estimates appear to be lower than present day values and can always be fully explained by abiotic chemical reactions to 98% (Tab. 7.33). The data leaves virtually no extra room (2%) for an additional process in addition to those we already know, and the need to invent new pathways is really very small. For our case study at Jämjö, we may make the following table showing our best estimate of the causes for potassium weathering under an Oak-Norway spruce mixed stand with feldspars, hornblende, epidote, biotite and vermiculite minerals in the soil:

| Dissolution cause | Amount released kg K ha ⁻¹ yr ⁻¹ | Contribution to the total |
|------------------------------------|---|------------------------------|
| Reaction with acidity | 2.402 | 33% |
| Hydrolysis with water | 2.184 | 30% |
| Reaction with organic acid ligands | 1.092 | 15% |
| Reaction with carbon dioxide | 1.456 | 20% |
| Total weathering rate | 7.280 | 100% |

The PROFILE model includes the reaction of hydrogen ions, organic acids and carbon dioxide, and to the degree that mycorrhiza act through these factors, it implies that the effect of mycorrhiza is accounted for in the model. If the action of mycorrhiza is considered to be through the action of exuded organic acids and hydrogen ions, then the contribution from mycorrhiza becomes an exercise in determining how much of the flux of organic acids which are caused by mycorrhizal exudation as compared to the flux caused by microbiological decomposition of organic matter in the soil. Mass balances suggest that the fraction of the soil acidity flux not accounted for by large macrobiotic causes or abiotic fluxes is fairly small. Model that ignore acidity exudation by mycorrhiza, which is not matched by cation uptake, do predict observed soil pH very well, supporting the view given above.

The contribution of mycorrhiza to the weathering was estimated using published values for estimates of hole production in stones, assuming all holes to be originating from fungal action. This way we can arrive at weathering rates in the magnitude of $0.15 \text{ keq ha}^{-1}\text{yr}^{-1}$ at Gårdsjön-like catchments. Earlier, we have made similar calculations for the catchments F1 and G1 at Gårdsjön, showing equally small residuals in need of explanation. In fact a diagram such as Fig. 4.1 indicate that the room for specific action by mycorrhiza to increase the weathering rate may be so small that it is undetectable in budget studies. The statement sometimes heard at conference and alluded to in the Nature article that "nearly all weathering is caused by mycorrhiza action", is systematically neglecting abiotic processes, systematically ignoring very much of available data, systematically ignoring published studies that show that the very basis for that hypothesis is unreasonable. It is well known that holes in mineral particles are abiotic phenomena connected to etch pits and crystal properties and they are observed in sites where no fungi ever was present. At best, we can conclude that such statements are very naive. Published photographs show that mycorrhiza can make small marks in stones, but a small bit of calculations show clearly that the total amounts released by this mechanism per hectare land is negligible.

Trees in control ?

The fourth myth cannot be found explicitly in the literature, yet is still persists among many foresters and geochemists, it generally runs like: "If there is too little base cations in the soil, then the trees will simply increase the weathering rate". The trees can modify the chemical environment somewhat around their roots, however, do not stand a chance of changing the bulk chemistry of any larger volumes of soil. For that, mass balances tell us that so large physical amounts of alkalinity must be employed that no tree has such amounts at its disposition. When averaged out over the whole volume enclosed by the roots, soil $\text{pH}(\text{H}_2\text{O})$ changes induced by trees are limited to maximum 0.2 to 0.5 units, and that also over substantial amounts of time. The soil solution organic acids may be increased by the tree under stress at the most by 5% to 20% in relation to the flux amounts normally moving. This is by far not significantly enough to change the weathering rate in the rooting zone by a few percent. A 50% increase in the weathering rate would require the organic acid concentration to rise by 10 to 30 times. It is questionable if the root would survive that, nor that it has the capability to something of the sort. A significant increase in organic acid output is costly to the tree, and for a tree in stress, this kind of expenditure may soon prove fatal.

Trees do not know where they are, nor can they send their roots to any particular location. The trees incessantly search the immediate vicinity with their roots in a "random walk" strategy, where new roots repeatedly grow into the soil under the tree. Any discovered favourable condition discovered this method is opportunistically exploited. Roots in unfavourable locations will wilt and appear as a cost in the tree system. In the case of overlapping root space with other trees, competition is the rule.

Thermodynamics is it ?!

The fifth myth is the largest of them all and involve the most prestige and potential for loss of face. Thus, resistance to revise it has been compact. The traditional soil chemistry models built by geochemistry specialists, originally developed for issues in agriculture and groundwater geochemistry, were wholly based in equilibrium processes. Most of them were first developed in the United States. This approach had historical reasons, before the advent of modern computers, systems of non-linear differential equations were mathematically insolvable for any practical use, and other less correct, but operable approaches had to be developed. These traditional models were initially operated as "equilibrium diagrams", later they were converted to code and enlarged, they rely much on large amounts of data for calibration and something called "thermodynamic databases". If this requirement is fulfilled, they will sometimes produce useful backcasting and can be used for making extrapolative predictions somewhat outside the area of calibration.

These models quickly run into problems without large amounts of data to calibrate on, and this is caused by a severe specific shortcoming that must be pointed out, even if many do not want to hear it. The traditional geochemical codes use equilibrium formulations for processes which are valid for soluble salts, carbonate dominated systems and other reversible reaction systems. But such models are beyond all doubt, not formally valid for kinetically controlled and irreversible reaction systems. This include all the soil mineral weathering reactions of practical interest here. The so-called "thermodynamic equilibrium databases" for soil and geochemical silicate weathering reactions referred to are not "thermodynamic" at all, they are at best steady state balance coefficients or empirical mass transfer coefficients that have been misinterpreted as equilibriums. Very little of the data in the "thermodynamic equilibrium dataset" will stand up to any formal inspection with respect to classical requirements for equilibrium. The term "thermodynamic" is thus very bogus in this context, if it was not, we should be able to precipitate feldspar and hornblende from aqueous solutions in the laboratory, but we can absolutely not ! Thus, such models may be useful for simulations or design work after careful calibration, but do not expect them to explain processes, nor to be particularly reliable. The fact is often overlooked, that without the excessive calibration, these model lack all capacity of any a priori predictions. Prediction field weathering rates from mineralogy and texture, or any other observable soil property, they cannot.

Many geochemists become very defensive when this issue is discussed, maybe seeing it as a threat to their profession. I think their blatant denial of the obvious and conservatism leading to clinging to a dead paradigm, is the real threat to the credibility of predictive geochemistry, and that the whole field of "thermodynamical equilibrium geochemistry" is in dire need of a grand revision and rethinking of its core paradigm.

Chapter 8

Forest vitality and stress implications

Changes in forest vitality affect such fundamental tree functions as growth, reproduction and defence against pests (McLaughlin and Percy 1999). Stress factors acting on the forest limit the resource utilisation, effective growth and reproduction (Grime 1991). For sustainable forestry in southern Sweden, the current objective is to balance the goal of achieving high growth per year, high wood quality, of preserving or increasing biodiversity, of maintaining long-term soil productivity and of preventing the leaching into streams of soil nutrients and metals due to natural and anthropogenic stress factors.

8.1 Mechanisms and systems of multiple stress

Stjernquist, I. and Sellén, G.

For several decades, forest damage involving various tree species in Sweden has been reported. Recently the focus has been on the increasing oak decline. During the early 1980s an increase in defoliation was also reported for both Norway Spruce and Scots Pine. The defoliation status of these two species has been subjected to systematic monitoring by the National Forest Survey since 1984, and with higher spatial resolution for various regions through air-photo interpretation since 1986. Beech and oak have been subjected to monitoring since 1988. Natural selection has made the indigenous tree species in Sweden well adapted to cope with a number of natural environmental stress factors, as well as the variation in these. Over the centuries, naturally regenerated forests have adapted to local conditions regarding the soil and other edaphic and climatic conditions. Stands planted on less suitable sites, such as the soil being too dry, too poor or too rich, can suffer from reduced vitality and through that from increased risks for fungal and insect attacks. The continuous and systematic monitoring of tree vitality in southern Sweden also show that, despite a intensive forest management, tree condition failed to improve over time. The major stress factors involved must either be anthropogenic or extreme natural events. In more specific terms what are the driving stress factors, how do they interact, and what is the relative importance of a specific factor for the growth and reproduction of forests? These questions can only be answered by a system analysis approach.

8.1.1 Interaction of stress factors

Larsen (1995) noted that the stability of a forest ecosystem is traditionally defined in relation to catastrophic events. These can include damages by wind and snow, as well as intense pathogenic attacks. However, a forest system is in continuous interaction with the surroundings. Thus forest health is also affected by long-term stresses of low intensity. From the perspective of forest health, the statement that ecosystem stability is a function of points to the importance

of physiological changes caused by long-term stress factors as that of ozone. A system analysis of stress factor interactions for tree vitality must try to identify not only the joint effects of the factors involved, but also the thresholds when visible damages appear (Manion 1991). In the process of events that give rise to an outbreak of pest or disease or to a sudden decrease in tree vitality, defined as growth, three groups of factors or agents can be identified:

- Predisposing factors
- Inciting factors
- Contributing factors

The predisposing factors affect trees over a long time period and contribute to a gradual increase in their sensitivity to external stress. The inciting factors, in contrast, affect trees by changes in conditions over much shorter, or even episodic, periods of time such as in the case of spring frosts or summer droughts. Contributing factors, in turn, may be insects or pathogens that have benefited by some earlier attack, by a summer drought or by a chronic root rot infestation that spreads within a stand through root or mycelium contact.

Predisposing factors

Waring (1987) has described the characteristics of trees that are predisposed to die. The crucial process is that of changes in carbohydrate allocation. Unstressed trees show a seasonality in growth and in production of storage reserves and defensive compounds. The hierarchy for carbon allocation is as follows:

- leaves
- roots
- storage reserves
- stem growth
- defense compounds

When trees are exposed to stress, the priorities can change, for example, in trees exposed to ozone priority is given to defence compounds (Skärby et al.1998). Compared with long-term stresses, pulse stresses may have only a minor effect on tree vitality due to the trees being able to replenish its storage reserves later in the season. Loss of foliage and changes in canopy structure caused by long-term stress periods, however, decreases the photosynthetic capacity of a tree and thus the production of storage and defence substances.

Nowadays, the long-term predisposing factors are mainly those of anthropogenic, global or regional air pollutants, and their secondary effects on soil productivity nitrogen and sulphur deposition increased ozone concentrations, climate change, soil acidification and an imbalances of nutrient concentrations in the soil. Only the two last factors can be counteracted within the forest ecosystem. Sub-optimal biotopes and oxygen deficits in the soil are predisposing factors that need to be counteracted by forest management methods. Predisposing factors, that have effects lasting for no more than a season are those of sub-optimal moisture and temperature. Through decreased carbohydrate production, these stress factors enhance the sensitivity of a tree to frost damage. Since for most Swedish forest soils nitrogen availability has traditionally been the limiting factor, enhanced nitrogen deposition can be expected to enhance tree production. With a resulting larger biomass, the demand for water to maintain growth and metabolism will increase, making the water availability in the soil more crucial. A continued high nitrogen deposition can eventually affect the forest ecosystems negatively through nutrient imbalance and reduced frost hardiness. Research both in Europe and North America has shown no, or a

negative, growth response to nitrogen both on the part of conifers and of broad-leaved species (Emmet 1999). Research has also shown that insect damage can increase significantly with a high nitrogen deposition (Flückiger and Braun 1999)

A reduced relative availability of base cations through an increased uptake and storage in the biomass as well as the increased leaching from acidified soils, make trees more sensitive to external stress factors. For example, lack of K, Ca and Mg leads to a reduction in winter hardiness. Similarly, heavy attacks by beech scale, *Cryptococcus fagi*, have been linked with low soil pH (Jönsson 1998). Also an imbalanced nutrition increase the susceptibility of trees to *Armillaria* root disease (Entry et al. 1986). In southern Sweden the base saturation of the soil is below normal, <20%, at most monitoring sites and the imbalance of essential nutrients to nitrogen has increased over time. (Sonesson 2000, Jönsson, U. et al. 2001).

As a predisposing factor, tropospheric ozone is of particular importance. Ozone reduces photosynthetic activity and has a negative effects the transport of carbohydrates, particularly to the roots (Skärby et al. 1998). A significant reduction in stem growth in Europe for beech has been documented since 1984/89 (Flückinger and Braun 1999). One explanation of this may be the current levels of ozone. Estimations of European forests at risk of decreased health, evaluated by the exceedance of the AOT40 critical level, give a figure of 72% of the deciduous forests and 41% of the coniferous forests (Hetteling et al. 1996). More rapidly growing species such as birch, poplar and aspen are more sensitive to an increase in ozone levels than more slowly growing species like spruce.

Weather conditions during the year affect the ability of trees to develop frost tolerance, since the development of frost hardiness requires the availability of carbohydrates. Any summer conditions that result in a reduction in carbohydrate production could thus in principle, lead to the development of poor frost hardiness. Major defoliation through insect attacks, deficiency or imbalance of nutrients and elevated ozone levels can have similar effects. Defoliation by insects decreases the starch content of the twigs (Warin 1987). Carbohydrates are of importance also for the development of defence against insects and other pathogens. An expected future rise in temperature, as a result of an elevation in greenhouse gases in the atmosphere is, depending on location and site conditions, likely to affect the vitality, competitiveness and survival of any given tree species. A temperature increase of a few degrees would affect the southern limit for the occurrence of Norway spruce. Today this limit runs along the northern border of Scania, but with a 2°C increase Norway spruce would be likely to recede to central Sweden (Bradshaw et al. 2000). A warmer and more moist climate as well as a climate with insignificant winter frost promotes population development for many insects and pathogens, thus increasing the risk of insect and pathogen outbreaks (Hofgaard et al. 1999). A higher winter temperature is also assumed to decrease the carbon storage in Norway spruce through enhancing winter respiration.

Inciting and contributing factors

The inciting factors can be weather extremes (frost and drought), insects, fungi or forest management methods. For example, the cold winter in 1987 has been viewed as an inciting factor for the development of the widespread oak decline. That winter, the temperature was 4°C lower than average during both December and January. A spring frost backlash in April to early May has also been reported to cause damage to the stem cambium and thus affect tree vitality. The increased resin flow on spruce stems in the early 1990s has been related to a severe frost backlash in 1991 (Barklund et al. 1995). Trees with a shallow root system have a higher risk of drought stress. Worrel (1983) reported dense stands of Norway spruce with shallow root systems being attacked by bark beetles due to the reduction in carbon storage following a drought period. The factors contributing to the death of trees are generally pest and pathogens that have been promoted by the inciting factors referred to or have chronically infected a forest stand, such as root rot or *Armillaria* root disease.

Forest management as a countermeasure

Although many of the predisposing factors may slowly reduce forest vitality, an appropriate forest culture and management may offset these effects to some extent. Natural regeneration, the presence of shelter-wood, mixed stands, choice of an appropriate species or provenance, removal of forest residues and the avoidance of stem and root damage during forestry operations can all have a beneficial effect. Investigations in southern Sweden indicate that mixed stands have a better potassium status than pure spruce stands (Thelin et al. 2001b). Mixed stands may also be better suited to withstand storm conditions due to an optimal root penetration of the soil

8.1.2 Memory effects

Many of the metabolic changes caused by stress factors, can persist for days or even months. For example, when Norway spruce and Scots pine seedlings were exposed to two to four times the ambient ozone concentrations for several weeks, the chlorophyll content of the 1-year-old pine needles at the end of the exposure period was found to have decreased (Langebartels et al. 1997). In spruce an ozone-dose-dependent decrease in chlorophyll was observed the next season. Memory effects have also been noted in birch, such as reduced contents of Rubisco, chlorophyll, starch and nutrients in the leaves and a decrease in new shoot growth (Oksanen and Saleem 1999). Memory effects may also occur after sub-optimal summer temperatures or an attack of foliage grazing insects, both factors decreasing the capacity of trees to sustain winter frost temperatures (Olofsson 1986). Changes in root/shoot ratio, crown structure, defoliation and mycorrhiza can also be regarded as memory effects, that can have long-term effects on forest health.

8.1.3 Forests management as a stress factor

The intensity of forest management in southern Sweden has changed during the last century. The traditional multiple use of forests, within the framework of traditional farming, resulted in open forests, often dominated by mixed species. Compared with a modern planted forest stand, the standing biomass per ha was low. The changes in preferred outcome from the forests caused a switch over to high productive monocultures of spruce, which increased the nutrient uptake during a rotation period. Falkengren-Grerup et al. (1987) estimated the decrease in soil nutrient content that occurred between 1947 and 1984 (Table 8.1). The changes were assumed to have resulted from an increase in soil acidification. In efforts to estimate the increase in nutrient uptake during a rotation period, entirely as a result of changes in the intensity of forest management, a comparison was made between a spruce forest from 1923 and an intensively managed spruce stand from 1980. The starting point was chosen to represent the old type of forests common prior to passage of the Swedish Forestry Act of 1918, a statute which for the first time aimed at protecting growing forests (Stjernquist 1973). The end point represents the forests conditions when a maximal biomass production was the goal for the Swedish forest policy, defined in the Swedish Forestry Act of 1979.

The first Swedish forest survey, which took place 1923-29 found the average timber volume per ha in southern Sweden to be $75 \text{ m}^3 \text{sk ha}^{-1}$ (total volume over bark). By 1980 the timber volume in the counties of Malmöhus and Kristianstad in the Scania region of southern Sweden had increased to $153 \text{ m}^3 \text{sk ha}^{-1}$ (Bengtsson et al. 1989, Miljöstatistik 1986-87). The assumption is that both stands are growing on clayey till in southern Sweden and therefore the nutrient content of different parts of the tree was taken from an investigation of the nutrient conditions in a 55 year old Scanian spruce forest (Nihlgård 1972). It was also assumed that the nutrient concentration in different parts of the spruce tree was the same in 1923 and 1980. The results are summarised in Tab.8.1. Compared to the leaching of nutrients caused by soil acidification, the negative effects of forest management were by no means insignificant. However, the major difference between these two factors is that whereas the leaching of nutrient constitutes

| Nutrient | Difference in nutrient uptake 1923-1980, kg/ha | Nutrient decrease in the upper soil layer 1949-1984, kg/ha |
|----------|---|---|
| N | 129 | - |
| Ca | 77 | 76-1765 |
| K | 72 | 16-203 |
| Mg | 11 | 11-137 |
| P | 15 | - |

Table 8.1: Increase in nutrient uptake in a spruce forest between 1923 and 1980 as compared with nutrient decrease in the root zone between 1949 and 1984 (Falkengren-Grerup et al. 1987)

a permanent loss to the ecosystem, the effects of an increase in biomass per ha depends on end-use.

8.1.4 Plant strategy and stress

The C-S-R model of plant strategies (Grime 1979; Grime et al. 1996) enables plant species to be classified with respect to resource capture and allocation and to competition in terms of response to stress and disturbance. Stress is defined by Grime (1979) as circumstances that limit photosynthetic production, such as shortage of light, water or mineral nutrients and temperature, whereas disturbance consists of the partial or total destruction of plant biomass, such as pathogen and herbivore damage, drought or soil erosion. All plant habitats are more or less affected by stress and disturbance. The survival and reproduction of plants depends on their ability to cope with these environmental changes, atmospheric changes included. Stress-tolerant (S) species has a low potential relative growth rate and their nutrient uptake is uncoupled from the vegetative growth. Carbohydrates and nutrients are stored in the leaves, stems and roots of such trees allowing them to survive for a longer period of time without nutrient uptake and dry-matter production. Their morphological responses to stress are slow and small in magnitude. Environmental variations tend to be counteracted by physiological rather than by morphological changes. Species that are competitors (C) have a high potential relative growth rate and their nutrient uptake is correlated with the vegetative growth. Most carbohydrates and nutrients are rapidly incorporated into the vegetative structure. Their morphological responses to stress are rapid, maximising vegetative growth. Typical reactions are changes in the root/shoot ratio and in the leaf or root area. Ruderals (R) have short life spans and are usually annuals or short-lived perennials. Between these three main types, there are intermediate strategies of all conceivable sorts. In terms of this classification system, Swedish tree species show mainly the the strategies of being competitors (C) or stress-tolerant competitors (SC), some species representing intermediate form (C-SC), Tab.8.2 (Grime et al. 1996). Within the C-SC and SC groups there are considerable variations in growth capacity. For example, birch has a relative growth rate close to that of the C group. Of the Swedish tree species classified as C-SC, birch has been found in aeroponic experiments under optimal nutrient conditions to have a high potential RGR, one of 27%. The potential RGR of stress-tolerant competitors, such as spruce and beech has been estimated to be 6-7% (Ingestad 1981, Möller Nielsen 1998).

The nutrient reaction

Stress-tolerant species are to be found in habitats with chronic nutrient deficiency, mainly of N and P. The nutrient cycling are closed, the decomposition of organic material being a key factor for nutrient availability. If the nitrogen input increases slightly, this results initially in

| Competitor (C) | Intermediate (C-SC) | Stress-tolerant competitor (SC) |
|--------------------|---------------------|---------------------------------|
| Salix ssp | Betula agg. | Picea abies |
| Fraxinus excelsior | Acer platanoides | Fagus sylvatica |
| Ulmus glabra | | Quercus robur and petrea |
| | | Alnus glutinosa |
| | | Sorbus aucuparia |

Table 8.2: *Swedish tree species, classified by the C-S-R strategy model defined by Grime (1979) and Grime et al. (1996)*

an increase in production (Tamm 1991) Since the major influxes of nutrients to stress-tolerant species growing on infertile soils come in pulses, an evenly distributed root system is a positive morphological characteristic for the individual, so as to maximise nutrient uptake and maintain a high vitality (Grime 1991). The nutrient transfer from the soil to the root surface is another key process, one that limits the availability and uptake of nutrients in this strategy class. To counteract this, stress-tolerant plants need to have a large root biomass, together with mycorrhizal infection, so as to maximise nutrient acquisition through exploiting a large soil volume (Chapin 1980). The storage of nutrients is a positive characteristic for survival, since stored nutrients can be utilised under periods of limited supply (Marschner 1997). To counteract a slight imbalance in nutrient availability, the storage capacity is an adaptation to stress. An excess of nitrogen in the soil, and the leaching of cations below the root zone, are stress factors, that are abnormal for stress-tolerant species. They create a continuous state of stress, one which differs from natural stress situations to which a stress-tolerant species has evolved. Nitrogen excess also changes the root morphology, the root/shoot ratio and the amount of mycorrhiza.

Strategy class and morphological characteristics

The effects of stress factors are not only influenced by the strategy class but can also be enhanced or moderated by the morphological characteristics of the tree species itself. Maintenance of the water balance, for example, which is a crucial process for tree vitality is affected by the junctions between the branches and the stem, which constrict the water flow. This has been shown for many species, such as for Acer, Betula, Populus and Fagus (Tyree and Evers 1991). A plant's reaction to limited water resources is to sacrifice minor branches so as to maintain the water balance (Zimmermann 1978, Tyree and Sperry 1988). For beech, water transportation is negatively correlated with the number of bud scars per unit branch length, which increases the effects of a water deficit. Stress factors, that change the crown structure and hamper the growth of lateral shoots cause a lasting reduction in water transportation. The response of the tree to sustain the transpiration fluxes is to lower the stomatal conductance, which decreases the length of the next year's shoots (Rust and Hüttl 1999).

Predicting stress reactions

To be able to predict the specific stress reactions of a given tree species in Sweden it is important to evaluate its life strategy. Typical stress-tolerators are adapted to unproductive habitats, their survival being possible through the rapid capture of resources under favourable conditions and the ability to store them for later use during unfavourable conditions. Typical competitors, on the other hand, are adapted to stable and productive habitats, their success depending less on storage than on their ability to maximise resource capture by means of rapid morphogenetic adjustments achieved through reallocation of captured resources. A stress factor, that results

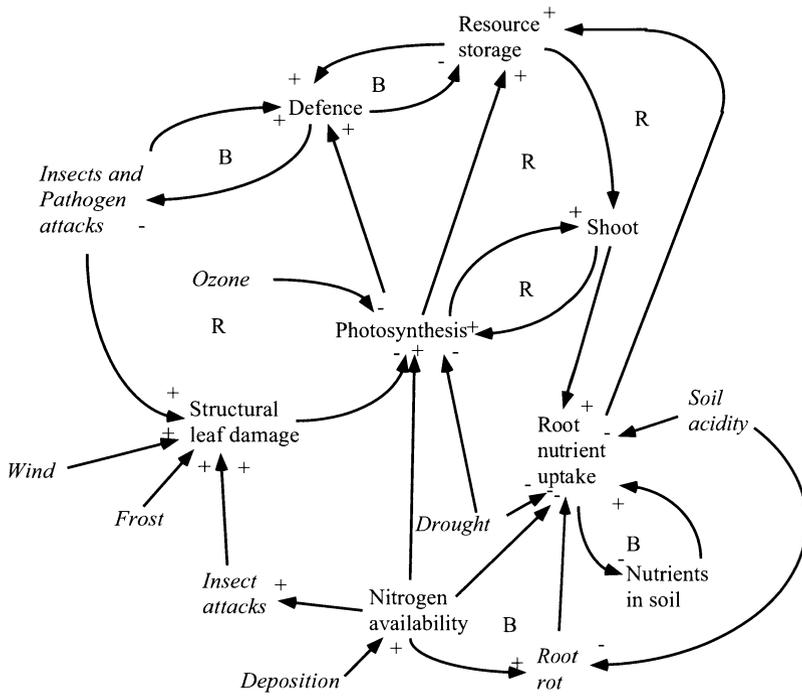


Figure 8.1: Causal loop for the tree system, linking growth, resource storage and different external factors causing damage and disturbance.

in a decrease in net photosynthesis may reduce growth. However, since stored resources can be used to compensate for stress, at least for short periods of time, species with strong stress-tolerating characteristics tend to be less sensitive than competitors. On the other hand, stress factors leading to leaf or needle loss are likely to reduce growth more in stress-tolerators than in competitors. Although both competitors and stress-tolerators are adapted to habitats in which the frequency of disturbances is comparatively low, the low growth rates of stress-tolerators does not enable needles or leaves to be rapidly replaced. On the other hand, a minor effect on growth of a stress-tolerant competitor does not mean that the effect of the stress factor is insignificant. Depletion of storage and defence substances may be as important, through decreased frost hardiness and defence against pest and pathogens. Models based on growth strategy and classification of the physiological characteristics of plants for predicting responses to stress factors have been tested. In an example with ozone, injury was estimated from both the Grime classification and the Ellenberg classification of plant species. Although both methods were useful, the Ellenberg classification, which includes the species reaction to light, water and temperature, performed better (Ball et al. 1999).

8.1.5 System analysis of the stress/health relationship

Many long-term predisposing factors affect the carbon pool and allocation within the tree. A system analysis of the relationship between stress and health of forests in southern Sweden shows that ozone, nutrient uptake capacity, drought and structural leaf damage have an early and joint effect on forest health through the changes in carbon allocation, Fig. 8.1. Nutrient

balance or imbalance may counteract or increase this effect, soil acidity and nitrogen availability being the most important agents. Carbon allocation influences both resource storage (structure and metabolism) and the production of defence substances. The chronic effects of the predisposing stress factors also cause changes in tree architecture, that lead to a decrease in potential carbohydrate production through a decrease in leaf area. The stress factors are also directly affecting the net photosynthetic capacity per leaf area. If tree vigour, expressed as wood growth per unit of foliage, decreases below a certain threshold value, the risk for pathogenic attacks and diseases enhance as this risk is determined by the amount of available defence substances in the tree.

8.2 Forest health indicators

Stjernquist, I., Rosengren, U., Sonesson, K., Sverdrup, H., and Thelin, G., Nihlgård, B.

Changes in forest vitality affect fundamental tree functions such as growth, reproduction and defence against pests (McLaughlin and Percy 1999). Stress factors acting on forest, limit the resource utilisation, effective growth and reproduction (Grime 1991). Forest vitality has to be studied and discussed both at the individual tree level - in order to better understand mechanisms, interrelationships and responses of the tree to stress - and at the stand and ecosystem levels to be more useful for management purposes. To identify the most useful forest health indicators, both the tree and the ecosystem levels need to be taken into account.

Various indicators of forest health or forest decline have been put forward and been used in different monitoring schemes (Alexander and Palmer 1999, Anon 1997a, Ferretti 1997, Innes 1993a, McLaughlin and Percy 1999). All such monitoring programmes aim at finding indicators, preferably in the field, that are easy and robust to measure as well as being able to clearly show the link between an agent and changes in forest health, ideally in terms of tree growth. Some indicators focus on the vitality of individual trees and others on ecosystem changes, for example the frequency and kind of pathogens, biodiversity, the C/N ratio in the forest floor (Gundersen et al. 1998) and the soil conditions as defined by the ratio of base cations to aluminium (BC/Al^{3+}). Forest health can also be investigated indirectly by assessing the cover of different tree epiphytes that are sensitive indicators of changes in environmental conditions, such as light, moisture, nitrogen deposition and air pollution (Ellenberg et al. 1992, Hultengren et al. 1991). Lichens and mosses are often more sensitive to environmental changes than trees and the species composition reflects the long-term impact of stress factors.

This chapter will evaluate existent indicators in relation to long-term growth and reproduction. The time and space aspects are important in this assessment. If the time lag between agent and effect is too long, the stress factor can be ignored or, as in the case of pest outbursts, the link between stress factor and effect is unclear. Besides being specific, indicators for general use need to be easy to measure for monitoring activity outside the research field as well as cheap to sample.

8.2.1 Visible indicators for assessing tree vitality

Defoliation indicators

For coniferous species, visible damage indicators in general use in Europe are discoloration and defoliation in percent, assessed in the field or from large-scale colour infrared aerial photographs (IR); the two methods are of comparable accuracy (Skogsstyrelsen 1994, Schlyter 1993, Schlyter and Anderson 1997). Monitoring personnel can be educated to evaluate changes in defoliation in a standardised way through quality assessment schemes, so that differences between individuals are insignificant for the outcome of the assessment (Innes 1993a, Hansen 1996). In a Swedish investigation of needle loss, the difference between individuals assessing the canopy was only 4% (Schlyter and Andersson 1997), well within the range of normal confidence intervals for e.g.

chemical analyses. Age, anthropogenic stress factors, insects as well as root and butt rot are the most important agents influencing the degree of defoliation (e.g. De Vries et al. 1997, Hendriks et al. 1994, Landmann et al. 1995). Despite being a subjective method, one knows that there is a clear relationship between defoliation and forest vitality in the sense that without needles there is no growth. There are, however, only a few studies that show a relationship between growth rate and needle loss for conifers in Sweden (Söderberg 1993), in North America (Barnard et al. 1990) and in the Vosges Mountains in northeastern France (Becker 1990). The Swedish study is based on needle loss estimation and coring for 16 650 Norway spruce trees and 15 000 Scots pine trees at equally many sites randomly distributed over the forest area of Sweden (Söderberg 1993).

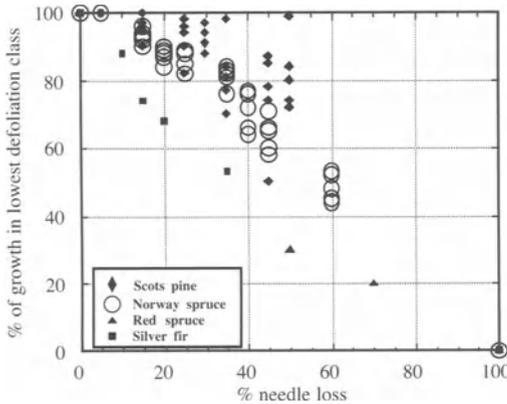


Figure 8.2: Relation between observed defoliation and stem growth for Norway spruce (*Picea abies*), Scots pine (*Pinus sylvestris*), Red spruce (*Picea rubens*) and Silver fir (*Abies alba*). Data from Barnard et al., (1989); Becker (1991) and Söderberg (1993).

stem growth in % of undamaged trees, has been proposed by Söderberg (1993). On the basis of graph 1, the following empirical relations between needle loss (BF) in % and stem growth r_G in % of undisturbed trees (100%) were obtained for Norway spruce, Scots pine, Red spruce and Silver fir.

$$r_G(\text{Norwayspruce}) = 101.9 - 0.575 \cdot BF - 0.0045 \cdot BF^2 \quad (8.1)$$

$$r_G(\text{Scotspine}) = 99.51 - 0.032 \cdot BF - 0.0096 \cdot BF^2 \quad (8.2)$$

$$r_G(\text{Redspruce}) = 99.94 - 1.667 \cdot BF - 0.0097 \cdot BF^2 \quad (8.3)$$

$$r_G(\text{Silverfir}) = 100.6 - 1.614 \cdot BF - 0.0061 \cdot BF^2 \quad (8.4)$$

The set of equations can be inverted to give the relationship between needle loss and growth

$$r_L(\text{Norwayspruce}) = 98.9 - 0.577 \cdot r_G - 0.0036 \cdot r_G^2 \quad (8.5)$$

$$r_L(\text{Scotspine}) = 98.76 - 0.252 \cdot r_G - 0.0063 \cdot r_G^2 \quad (8.6)$$

$$r_L(\text{Redspruce}) = 100.2 - 1.840 \cdot r_G - 0.0084 \cdot r_G^2 \quad (8.7)$$

$$r_L(\text{Silverfir}) = 100.0 - 1.519 \cdot r_G - 0.0053 \cdot r_G^2 \quad (8.8)$$

where r_G is the rate of growth in % of normal growth and NL is needle loss in %. The equations can be used to convert growth to needle loss or vice versa. For example, when the function for

basically, the same correlation was found for the five geographical regions defined by the National Forest Survey, despite large variations in climate within the surveyed area. This appears to exclude a bias due to climatic influences and to indicate there to be a fundamental relationship between tree growth and needle loss. In North America, a fewer number of trees were surveyed, a sample of approximately 2000 red spruce trees located in New England being involved (Krahl-Urban et al. 1988), Fraser fir appearing to display the same type of basic response. (Krahl-Urban et al. 1988). The French survey included a number of stands in small areas of NE France. The sample size for Silver fir was one of 1000 trees distributed over approximately 275 sites. The relationships obtained are shown in Fig. 8.2. For Norway spruce and Scots pine, the relationship between defoliation in % and

(Sonesson 2000b). Fig. 8.4 shows the relationship between needle mass and growth as a causal loop diagram. As can be seen, the foliar mass can be affected in several direct or indirect ways. Water, insects, ozone and frost can directly affect foliar mass. Drought and an excess of nitrogen, on the other hand, have an indirect effect through changes in the uptake capacity for nutrients, whereas the effect of butt rot involves several steps, affecting the architecture and efficiency of the root apparatus. Indirect effects in several steps will cause a significant delay between agent and foliar loss, thus being one of the reasons for why defoliation has been difficult to attribute to specific events.

Root vitality and root decline

The information available indicates there to be a coupling between plant survival rate and root growth reductions (Abrahamsen 1984, Ryan et al. 1986a,b, Keltjens and van Loenen 1989). Data on conifers has been plotted in Fig. 8.5, showing the pattern for Norway spruce, Douglas fir and Western hemlock to be basically the same. For Norway spruce a reduction of root growth of 55% is coupled with a 50% survival. In the long-term perspective, a 30-35% decrease in root growth leads to a seedling mortality of 100%. From data presently available the following empirical relationship for tree survival in % can be obtained:

$$Survival = -104 + 3.638 \cdot r_G - 0.0163 \cdot r_G^2 \quad (8.9)$$

where r_G is root growth as percentage of control. The relationship applies to the survival of young trees but experimental data of the same kind for mature trees is not available. However, it is possible that a similar relationship may also exist for mature trees. Support for this is found in the fact that older trees seem to suffer more from acid rain than younger trees. Although the roots are highly important for tree vitality, relatively little effort has been directed at investigating tree roots. However, several methods has been suggested where root vitality are used as forest health indicators. For instance, root vitality can be studied by classifying the roots with their associated mycorrhiza in different classes based on morphological structures (e.g. Clemensson-Lindell and Persson 1994). Nutrient and aluminium concentrations in the roots have also been used as indicators (e.g. Persson and Ahlström 1990/91) as have different physiological measurements of enzyme activity such as the TTC-method or ATPas activity (Clemensson-Lindell 1994, Rosengren-Brinck et al. 1995). Root biomass and root length can be estimated by extracting the roots from soil cores or using in-growth cores where mesh bags containing for example sand are placed in the soil so that roots and mycorrhiza can grow into it (e.g. Majdi and Persson 1993, Vogt and Persson 1991). The so-called minirhizotron technique can provide an insight into root dynamics without use of destructive sampling, providing novel information on root nativity and mortality (Majdi et al. 1992). However, although suitable for scientific purposes, most of these methods are very time-consuming and/or expensive, making them unsuitable as indicators of forest health for general use.

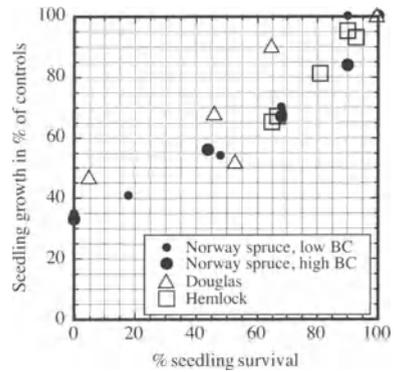


Figure 8.5: The relation between survival rate and root growth for Norway spruce in the experiments by Abrahamsen (1984), for Douglas fir in the experiments by Ryan et al., (1986a,b) and Keltjens and van Loenen (1989), for Western hemlock in the experiments of Ryan et al., (1986a,b) and Keltjens and van Loenen (1989).

Tree growth as an indicator

As discussed in chapter 8.1, many tree species in Swedish forests are stress tolerants with slow growth rates (Grime and Campbell 1991). These species, such as Norway spruce, show a slow morphological response to stress, the visible effects of stress factors on stem growth being difficult to identify if the monitoring period is too short. Since Norway spruce is such an important forest species in Sweden, many of the earlier studies of forest damage focused on spruce. In these studies, generally no effects of stress on stem growth have been found. This does not necessarily imply, however, that the stress factor under consideration has no effect on tree vitality. It is known, for example, that an indicator such as needle weight can respond to a change in nutrient availability within a single growing season and that an increase in needle weight correlates well with an increase in stem growth but it may take years for the increase to be detectable (e.g. Timmer and Morrow 1984). In assessment of tree crown vitality, there are other methods that have been used in combination with, or as an alternative to, needle loss and discoloration of foliage. These include for example the incidence of stress shoots and dwarfed shoots (Liedeker et al. 1988), a damage index based on the number of needle-age classes, discoloration, shoot length and the amount of dead branches (Tichy 1996), as well as the recently proposed branch development method (Thelin et al. 2001b). This method is based on the influence of tree nutrition on the allocation of dry matter to the branches (Madgwick and Tamm 1987, Flower-Ellis 1993). Thelin et al. (2001b) found relationships in spruce between branch length, branch needle mass, needle density, the shoot multiplication rate and the needle nutrient status several years earlier (approximately at the onset of growth of investigated branches). In addition, branch length was well correlated with growth, calculated as basal area increment, and with conventionally assessed needle loss. A limited nutrient supply will probably have a long-term negative effect on shoot development, whereas drought one year restricts shoot growth and bud development primarily during that particular year. During the years following increasing shoot development and growth may repair the damage done during the drought year, provided that the nutrient supply is sufficient. Indicators such as branch length and the shoot multiplication rate may be appropriate vitality indicators in the assessment of nutrient stress effects, whereas needle loss is a more general indicator of current tree vitality. The branch development method has the advantage of employing measurements rather than estimates. Another advantage is that indicators such as branch length and the shoot multiplication rate can include effects that have accumulated for several years, whereas needle loss and discoloration may be short-term transient phenomena. An approach to the stress/growth relationship involving statistical methods similar to those used in epidemiological studies in human medicine are useful for identifying small changes over long periods of time. Such an analysis of the relations of current ozone concentrations to spruce growth is described in chapter 8.4. The relations discussed above show that changes in stem growth over time is not a very useful indicator for monitoring continuous and low-level stress. If long-term experimental data sets exist, the epidemiological approach may be useful for evaluating the effect of environmental changes that directly affect the tree, such as ozone. The use of branch growth and foliar mass rather than stem growth, decrease the drawback of a long-term lag.

8.2.2 Nutrient concentrations and ratios to N as vitality indicators

A key type of indicator for Norway spruce and Scots pine used in Scandinavia is based on foliar analysis of essential nutrient and on the ratio of nitrogen to other nutrients. Foliar nutrient concentrations are closely related to growth (i.e. Lyr et al. 1992, Walker 1991), and have been used internationally as a diagnostic parameter for assessing forest health and changes in tree vitality over short periods of time. The critical level for deficiency is defined as the concentration at which growth is reduced by 10% (Ulrich 1952). As a diagnostic tool, the use of foliar chemistry for individual elements has the drawback that the concentrations can vary due to such factors as the nutritional status of the soil, the age of leaves, the position of the leaves

| Element | Deficiency | Parameter | Target value |
|---------|-------------|-----------|--------------|
| N | 12-13 | | |
| P | 1.1-1.3 | P/N | 10 |
| K | 3.5-4.0 | K/N | 35 |
| Mg | 0.4-0.7 | Mg/N | 4 |
| Ca | 0.4-2.0 | Ca/N | 2.5 |
| Zn | 0.008-0.015 | Zn/N | 0.05 |
| Cu | 0.002-0.003 | Cu/N | 0.03 |

Table 8.3: *Deficiency levels for nutrient concentrations and ratios in current year needles of Norway spruce, estimated from Scandinavian spruce stands. (Braekke 1994, Linder 1995, Rosengren-Brinck and Nihlgård 1995, Ingerslev 1998, Thelin et al. 1998)*

within the crown and the time of the year (Innes 1993a, Raitio 1995). Many of such drawbacks can be avoided, however, if the foliar nutrient content in proportions to nitrogen is used.

Table 8.3 shows deficiency levels in Scandinavia for individual elements in current year needles of Norway spruce, together with target levels for ratios to N (Braekke 1994, Linder 1995, Rosengren-Brinck and Nihlgård 1995, Ingerslev 1998, Thelin et al. 1998). The underlying assumption is that concentrations and ratios below target values indicate there to be nutrient imbalance in the tree. Ericsson et al. (1995a) has shown that when the P/N ratio is below 8 the arginine concentration in the needles increases, indicating an excess of nitrogen. K/N ratios below the target value may affect stem increment growth. For spruce stands older than 40 years, a K and P concentration in current year needles below the deficiency level, as well as a K/N ratio below 35, reduce branch growth by about 10%. Significant correlations have also been found between branch growth capacity and stem increment (Thelin et al. 2001b).

Most studies have been carried out on conifers, no deficiency or target values has been specifically tested for beech or oak growing in the poor soils that are dominating in Scandinavia. Linder (1995) has suggested, however, that all higher plant species have the same nutrient values for optimal growth. If this is the case, stress monitoring of deciduous tree species in different parts of Europe can use the same target values as conifers, Table 8.3.

Bergmann (1988) has provided optimum nutrient values for deciduous tree species growing in central Europe. If the values for beech are compared to Swedish data obtained at the end of the 1960s, i.e. before the period of high pollutant deposition, all the essential nutrients, except N, are well above the Swedish concentrations, even if the investigated beech stands are growing in nutrient rich soil, a clayey till. (Nihlgård 1972, Nihlgård and Lindgren 1977). In a later investigation of the nutrient content of beech stands growing on brown forest soils to acid humus-rich soils, Balsberg-Påhlsson (1989) found P and K concentrations in the upper crown foliage to be low as compared with the values given by Bergmann. Of the 22 stands investigated, 15% had a N/K value indicating there to be nutrient imbalance. Thus, more work is needed to verify the critical levels for deciduous trees in Scandinavia

However, there seems to be a clear connection between nutrient status and tree vitality. In an investigation of beech vitality in southern Sweden, the foliar K, N and P concentrations during July to August were negatively correlated with damage class, $P < 0.05$, Table 8.4. Compared with the values for conifers, the undamaged deciduous trees had higher N and K and lower P foliar concentrations. A graphical vector analysis of leaf mass, nutrient concentration and content for the three structural classes show a decreased availability of K, P and N for trees belonging to damage class 2 and 3. The branch growth and development during the last 6-7 years are to 50P/N ratio in July and August and by the K/N ratio in July (Stjernquist et al.

| Nutrient | units | date | IR class 1 | IR class 2 | IR class 3 | p value |
|----------|-------|-------|------------|------------|------------|---------|
| N | mg/g | June | 20.7 | 19.7 | 18.1 | 0.01 |
| N | mg/g | July | 19.8 | 19.1 | 17.2 | 0.001 |
| N | mg/g | Aug. | 21.2 | 19.4 | 17.1 | 0.0004 |
| N | mg/g | Sept. | 16.7 | 15.8 | 13.3 | 0.03 |
| P | mg/g | June | 1.17 | 1.12 | 0.98 | 0.0002 |
| P | mg/g | July | 1.07 | 1.05 | 0.94 | 0.01 |
| P | mg/g | Aug. | 1.12 | 1.06 | 0.89 | 0.004 |
| P | mg/g | Sept. | 1.17 | 1.10 | 0.74 | 0.0005 |
| K | mg/g | June | 5.24 | 5.05 | 4.12 | 0.07 |
| K | mg/g | July | 5.45 | 5.02 | 3.89 | 0.05 |
| K | mg/g | Aug. | 6.41 | 5.88 | 3.72 | 0.009 |
| K | mg/g | Sept. | 6.34 | 6.15 | 3.60 | 0.02 |

Table 8.4: *Nutrient concentrations in beech leaves from trees classified in different damage classes. The samples were taken from the upper part of the crown. n=46*

2002). For Norway spruce in southern Sweden, Thelin et al. (2001a) showed that the needle K status in trees older than 40 years was positively related to tree vitality, as measured by branch length and needle density, whereas in younger spruces tree vitality was limited by N or by N and P.

Nutrient imbalances in the tree change the carbon allocation, and thus indirectly affect the defence against frost and pathogens. Nutrient imbalance decreases the amounts of phenolic compounds, which are a strong defence against pathogen attack (Entry et al. 1991, Waterman and Male 1994). Sensitivity to frost, bark lesions and bark necrosis also appear to be correlated with nutrient status. Beeches that receive excessive amounts of N show an increase in the number of bark lesions and in frost sensitivity, as measured by the frost damage index (De Kam et al. 1991, Jönsson 2000a,b). A decreased K/N status in the tree has also been shown to stimulate infection by *Armillaria* (Moore et al. 1993).

These relationships show that foliar nutrient concentrations, especially the nutrient ratios to N, are directly connected with tree growth and vitality. Of the nutrients, N, K and P appear to be the most important although the deficiency concentrations and ratios to N appear to differ according to species, age and geographical region. As indicators of forest health for general use, foliar nutrient concentrations and ratios to N have the similar positive characteristics as defoliation but are more specific in linking the stress factor to an effect on tree vitality.

8.2.3 Soil chemistry

Since plants and the soil are intimately linked with each other, soil chemistry does have a great impact on plant performance. It is known that plants growing in nutrient rich soils have a higher nutrient content than those growing in nutrient poor soils. It is still unclear, however, how close the coupling between the plant and the soil really is. The question of whether concentrations in the soil *per se* or ratios of one element to another (e.g. BC/Al ratio) should be used It has also been discussed.

In studying changes in soil chemical status over both time and space, it is also important to have reliable reference values to relate to. Unfortunately, few attempts have been made to establish target values for forest soils. The natural variation is generally very great and may

| pH (H ₂ O) | pH (BaCl) | Ca | Mg | K | B | Cu | Zn | Base sat* % | C/N** |
|--------------------------|--------------|----|----|----|---|----|----|----------------|-------|
| 4.5 | 4.2 | 50 | 10 | 20 | 1 | 1 | 1 | 20 | 25 |

Table 8.5: *Suggested minimum reference values for pH, CEC, base saturation and concentration of various elements in the 20-30 cm soil layer in coniferous forest in southern Sweden. The data refer to analyses made d to principles used in ICP-Forest. * Maximum value for Al is suggested to be 100 mg/kg when base saturation is below 20%. **; Gundersen et al. (1998).*

be 10-100 fold over short distances. However, deeper in the soil profile the variation tends to be less. In Table 8.5 estimated minimum reference values for element concentrations in coniferous forest soils are introduced. These recommended minimum values are based on several hundred soil analyses taken from more than 100 sites in southern Sweden (south of 61 degrees latitude) during the 1980s and 1990s. All the sites were covered with conifers (Norway spruce and Scots pine), their age varying from quite young stands to old, mature ones. The soil types ranged from acid cambisols to well developed podsols. These values may be useful when considering the use of soil chemistry as indicators of long-term forest productivity and soil buffering capacity in order to ensure high quality run off water (i.e. no leaching of nutrients or heavy metals to surrounding ecosystems). The actual values given can be discussed but these recommended minimum values must be considered to be low for temperate forest ecosystems.

It is well known that under some conditions inorganic Al is toxic to plant roots. Aluminum chemistry depends on temperature, pH, parent material, ionic composition, humic and organic substances but due to the complex chemistry of Al it is still rather unclear under what precise conditions this happens (Schöttelndreier 2001). As an attempt to estimate when Al is toxic the ratio of base cations (mainly Mg²⁺, K⁺ and Ca²⁺) to aluminium (BC/Al ratio) has been widely used as an indicator for root vitality (e.g. Sverdrup and Warfinge 1993).

Soil chemistry is important not only from a long term forest productivity perspective but also in view of soil water seeping down through the soil profile to the ground water or the risk of contaminating the water in surrounding ecosystems. Due to the large heterogeneity of the soil, close monitoring of the soil water chemistry requires using vast number of lysimeters if reliable data are to be obtained. Use of the C/N ratio for the organic layer appears to be a promising new way to estimate the risk of NO³⁻ leaching from coniferous forest soil (Gundersen et al. 1998). If the C/N ratio is above 30, the risk for NO³⁻ leaching is very small, whereas when it is 25-30 the risk increases. When the C/N ratio is below 25 the risk for NO³⁻ leaching is considerable in stands with a closed canopy and no exponential tree growth.

8.2.4 Pathogens and insects as vitality indicators

Pathogens affect both tree and ecosystem vitality. While it is well known that strong outbreaks of butt rot and bark beetle severely damage forest production, the vitality of the trees is also influenced by the size of pest populations over time. Over the last two centuries, insects and pathogens have been investigated as a forest management practise as the change to monocultures has promoted the population growth of many pest species. The identification of large pest populations has caused practical forest management activities but has not been used appreciably in risk analysis.

Some examples of population changes in foliar grazing species due to the enhanced nitrogen availability through deposition or fertilisation have been reported. Increased nitrogen availability increase the infestation by shoot and leaf parasites, i.e. *Botrytis cinerea* and *Cinara pilicornis* in spruce (Flückinger and Braun 1999). Hoffman (1995) has estimated the effect on

tree growth of insect grazing, a population eating 1.5 kg, alternative 3 kg, foliage per day causes a 4% and 10% defoliation, respectively during the vegetation period. Consequently, beech stem growth was reduced by 2.6% and 5%, respectively.

The number of individuals in an insect or pathogen population is, basically, affected both by changes in weather and by changes in food resources during the year. To be able to use the size of pest populations as forest vitality indicator, the suggested species needs to be specific as well as linked with anthropogenic environmental changes. It is important to be able to separate the effects of forest management from the influence of other stress factors. The increase in population must also be easy to measure at an early time. The more or less stochastic nature of outbursts of pests and pathogens makes them often unsuitable for use as general indicators of forest health, although it is of vital importance to increase our knowledge concerning the processes and mechanisms why trees are attacked by pests and pathogens.

8.2.5 Evaluation of vitality indicators for trees and ecosystems

The vitality indicators most commonly used internationally for trees and forest stands are those of defoliation, crown structure, stem growth and nutrient status. Indicators for risk evaluation of the forest ecosystem include those of insects, pathogens and the ratio of base cations to aluminium (BC/Al) in the soil water. The discussion in this chapter shows that the vitality indicators most commonly used can be grouped in terms of their temporal and spatial characteristics

1. Indicators of tree vitality

- (a) A relatively clear relationship between the agent and tree vitality, the response being relatively fast,
 - i. easy and cheap to measure - e.g. defoliation, crown structure, nutrient uptake
 - ii. complicated or expensive to measure - e.g. root vitality, occurrence of mycorrhiza, changes in cell structure, frost sensitivity index
- (b) A relatively clear relationship between the agent and tree vitality, with a long time-lag before a measurable effect appears,
 - i. easy and cheap to measure - e.g. stem growth
- (c) A diffuse relationship between the agent and tree vitality, with a relatively quick response
 - i. easy or cheap to measure - e.g. bark lesion or resin flow
 - ii. complicated or expensive to measure - e.g. insect and pathogen outbreaks

2. Indicators of Ecosystem vitality

- (a) Indicators with diffuse relationship between the agent and tree vitality, with a relatively immediate measurable effect,
 - i. easy or cheap to measure - e.g. BC/Al in runoff water,
- (b) Indicators with a diffuse relationship between the agent and tree vitality, with a time-lag before a measurable effect
 - i. easy and cheap to measure - e.g. C/N ratio in the forest floor layer, epiphytic index

Although use of defoliation or crown structure as an indicator can have a relatively clear cause/effect relationship, in multiple stress situations many of the stress agents, in working together, can have the same overall result. In this situation, we therefore suggest nutrient

status is the best early warning key indicator of anthropogenic stress in trees and forest stands. Root vitality, mycorrhiza occurrence, changes in cell structure, etc. are just as precise but the methods used today are either expensive or time-consuming. Ecosystem indicators such as BC/Al ratio and C/N in the organic layer may also be useful, since they reflect the general state of the forest ecosystem in a broader landscape perspective.

8.3 Sensitivity to frost and drought

Anna Maria Jönsson, Niels Torkel Welander

Frost and drought sometimes cause damage to seedlings and trees in southern Sweden. Seedling establishment, needle functioning and plant growth are frequently affected. Moreover, frost and drought are predisposing and triggering factors for such damage as bark lesions on tree stems. However, frost and drought alone are usually not the only cause of damage, since they also interact with other factors such as light, temperature, wind, air pollutants, CO_2 and nutrient status of the soil and plants.

8.3.1 Frost

Background

This chapter will focus on frost damage that causes bark lesions, visible signs of wounds in the cambium. Bark lesions can affect wood quality and tree growth. Pathogenic insects and fungi invade lesions that are not healed, killing the tree in extreme cases. For instance, frost damage is considered to be the primary factor that favours infections by *Nectria spp.*, which causes bark canker (Day and Peace 1934, Chira and Chira 1998). During the 1980s and 1990s, bark necrosis and resin flow in Norway spruce appeared in southern Sweden to an earlier unknown extent (Barklund and Wahlström 1998). Bark lesions were found on 3% - 5% of the oaks and on 6% - 7% of the beech trees in 1988 and in 1993, respectively (Sonesson 1998, 1999a). Twenty percent of the declining oaks in northern Germany that were investigated were found to have bark lesions caused by frost damage (Hartmann and Blank 1992).

Sensitivity to frost

Frost damage usually results from a rapid drop in temperature to below zero following a warm period in the spring or autumn, or following a mild period during the winter. Spring frost is especially harmful when the trees have started to deharden (Schoeneweiss 1975, Thomas and Hartmann 1992, Thomas and Büttner 1993). One cause of bark necrosis and resin flow in Norway spruce has been suggested to be unusually mild winters, which can cause the trees to deharden early and thus readily be damaged by frost later in the spring (Barklund and Wahlström 1998). Mild winters may also cause trees to respire during the dormant season, which lowers the carbohydrate concentration and increases the risk of resin flow (Barklund et al. 1995).

Factors interacting with frost

Many factors influence sensitivity to frost of trees and make them more vulnerable to frost damage and to bark lesions. The sensitivity to frost differs among tree species and tree provenances. Beech, *Fagus sylvatica*, is considered to be one of the tree species most subjected to bark lesions (Day and Peace 1934). Beech has a smooth, thin bark

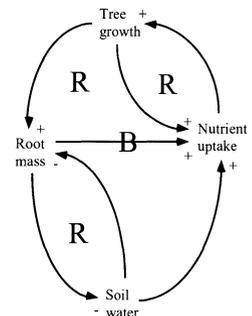


Figure 8.6: Causal loop diagram showing the effects of drought on tree growth.

without any adaptation to avoid strong fluctuations in temperature.

This contrasts to tree species with fissured and scaly bark that shades and insulates the inner parts (Nicolai 1986). Different provenances of the same tree species can differ in their sensitivity to frost. A provenance that starts to grow early in the season and grows for an extended period during the autumn is subjected to frost damage to a greater extent.

Oaks weakened by repeated defoliation, an excess nitrogen supply or locally by water stress are predisposed to frost damage, leading to their being attacked by pathogens (Kowalski 1991, Hartmann and Blank 1992, Kowalski 1996, Sieber et al. 1995, Thomas and Blank 1996). Not only mild winters, but also summer droughts are considered to trigger the formation of bark lesions with accompanying resin flow in Norway spruce (Barklund et al. 1995).

High concentrations of air pollution, as indicated by epiphytic lichens on the tree stems, increase the frequency of beech bark lesions (Jönsson 1998). High amounts of nitrogen can be a predisposing factor for frost damage through stimulating growth, delaying the hardening processes and lowering carbohydrate concentration (Burke et al. 1992, Larcher 1980, Nihlgård 1985, Skeffington and Wilson 1988). Within SUFOR, the effects of excess nitrogen on sensitivity to frost and the occurrence of bark lesions have been investigated. Spruces fertilized by an excess supply of ammonium sulphate were found to be more sensitive to severe frost treatment (Jönsson 2000a). Beech trees fertilized by excessive amounts of N had a significantly greater number of old lesions, the trees that had bark lesions seemed to be less able to withstand low temperatures (Jönsson 2000b). Although a negative influence of N fertilization could be traced in the concentration of nutrients in the bark as much as seven years after treatment, the absence of new lesions indicated that the vigour of the trees has increased. Trees having lesions showed higher concentrations of N and Al, indicating the occurrence of lesions to be correlated with soil acidity (Jönsson 2000c). Beech trees on soils with a relatively low C/N ratio and with low concentrations of mineral nutrients in the mineral soil layer were predisposed to be more frost sensitive (Jönsson 2000d).

Prediction of frost

The probability of a tree being damaged by frost can be predicted from tree and site properties together with data on climate. To avoid frost damage when regenerating a forest stand, it is important to choose a tree species and a provenance that are adapted to the local conditions. Vitality fertilization that counteracts nutrient imbalances in the tree can decrease the sensitivity to frost and the occurrence of resin flow and of bark lesions (Jönsson 2000a).

8.3.2 Drought

Background

The water potential of a plant may decline when water loss from it is larger than the uptake of water from the soil. This is the case when water availability in the soil declines. Plants and trees may then experience stress. Mild stress simply reduces growth but causes no damage, whereas severe stress may result in permanent damage. In the most extreme case, the plant may die. The impact of the stress depends on the severity of the soil drought i.e. the water content of the soil and the length of exposure as well as the number of drought events. Not only low soil water content but also low soil temperature can produce a water stress in the plant. This is especially true when the soil is frozen and when plants are exposed to high light intensity in the spring, causing water loss from the leaves or stems that water uptake by the roots is unable to replace. The influence of water availability on plants has been investigated extensively, such as in arid areas (Fisher and Turner 1978), and in trees (Hinckley et al. 1978, Dickson and Tomlinson 1996). That water availability may limit growth in Swedish forests is indicated by the fact that irrigation significantly increased growth (Nilsson and Wiklund 1992, Bergh et al. 1997).

Sensitivity to drought

Various processes in the plant differ in the degree to which they respond to a given level of stress caused by drought. Cell enlargement is inhibited by drought at a lower level of stress than many other metabolic processes in the plant (Hsiao 1973, Kramer and Boyer 1995). A plant is especially sensitive to low water availability when it is growing, e.g. when the shoots, leaves and roots are expanding (Tesche 1992). Following a drought, the shoots become short and the leaves small. The growth allocation also changes, so the plant endeavours to increase the uptake of the limiting resource, in this case water. Thus in plants under water stress root growth is favoured over shoot growth (Kramer and Boyer 1995) as has been shown in seedlings of oak and beech (van Hees 1997). A drought period may have a long-lasting effect, either during a growing season, or between seasons, as in the case of spruce (Tesch, 1992) or of beech, (Löf and Welander, 2000). One explanation for this may be that plant leaf area is reduced, resulting in a reduction in photosynthetic capacity. Such long-lasting effects can make it difficult to define how much of the growth reduction, or the damage can be ascribed to the current environmental conditions, since part of the reduction may be explained by a previous drought.

To assess whether the soil water level has caused a stress to plant growth, the relationship between growth and the soil water potential need to be known. In this context, it would also be useful if a close correlation between the soil water potential and growth rate of a plant could be established over a wide range of soil water potentials. However, with present knowledge it does not seem possible to provide a useful description of such a relationships, since knowledge here is restricted to a limited number of species and environmental conditions. At a water potential of about -1 - -2 MPa in the soil, growth has ceased in most cases, although the specific level differs between species (Kramer and Boyer 1995). According to Larson and Whitmore (1970), the leaf area and shoot expansion declined by potentials of about -0.2 MPa and root growth by about -0.4 MPa in *Quercus rubra*. In *Pinus radiata*, a significant growth reduction was observed at -0.05 MPa although root growth was reduced to a greater extent than shoot growth (Squire et al. 1987). Defining a critical level of soil water potential for a given species may be difficult since the roots of a plant, or a tree may penetrate to a considerable depth in the soil and supply the plant with water from alternative soil layers. Deep roots may supply shallower roots with water from deeper layers by a process called hydraulic lift (Horton and Hart, 1998). Moreover, water may be retranslocated within a plant (Kramer and Boyer 1995). Since the water potential of a plant is dependent not only on water availability in the soil but also on the transpiration rate, taking account of the atmospheric conditions is important in efforts to explain the influence of soil drought on plant growth.

Sensitivity to drought may vary between species. Seedlings of oak (*Q. robur*) are more drought tolerant than those of birch (*Betula pendula*). This has been explained by a deeper rooting and higher root growth rate at low water availability in oak than in birch (Osonubi and Davies 1981). A similar explanation was provided by van Hees (1997), who found oak to better tolerate drought than beech, due to an increase in root investment while maintaining its capacity for light interception when water availability declines. The above mentioned drought effects concern reduction in growth only. A different rating of sensitivity to drought between species might be found when drought is related to damage. In assessing drought tolerance by measuring the ability to withstand loss of water from the leaves, up to the point when the leaves become damaged, the following order of tolerance was observed, beginning with the most tolerant: *Pinus sylvestris* > *Picea abies* > *Prunus avium* > *Fagus sylvatica*, the least tolerant being *Quercus robur* (Lyr et al. 1992). Thus, differences between species in reaction to drought apparently depend on what plant effect is used as a measure of drought tolerance. It could also be expected that the death of leaves would lead to a reduction in growth, although this effect might not be detectable before the following growing season.

Factors interacting with drought

Various factors can interact with drought. Thus, light, temperature, wind, CO_2 and nutrients may influence plant water stress by affecting the loss of water from the leaves through either by nullifying or intensifying the negative effect of low water availability in the soil. Moreover, air pollutants, herbivory, availability of light and nutrients may affect growth, either by inhibition or by restriction through deficiency in addition to the negative effects of drought. Under high light compared to low light conditions the effect of a drought may be more severe, as in the case of oak and beech (van Hees 1997). A possible explanation for this could be that under high light conditions the transpiration rate is higher (Welander and Ottosson 2000), causing a greater stress to the plant. In oak seedlings subjected to a combination of drought and simulated winter browsing the interaction between drought and browsing was found to be small (Kullberg and Welander 2000). In this case the seedlings were mostly affected by the drought, since they were able to compensate for the loss of shoot tissue caused by browsing. In contrast to light and grazing, elevated CO_2 counteracted the negative effects of drought in seedlings of *Quercus petraea*, whereas drought at a normal CO_2 level reduced the growth of the seedlings (Guehl et al. 1994). Drought is sometimes used as an explanation for the failure of forest regeneration and the reduction in wood production. In such cases, the drought has not been manipulated to various desired levels but rather the water content, or water potential, has been measured and a certain level has been considered as representing a stress level or drought. In addition, it is difficult to assess how strong an effect drought has in forest since various other factors interact simultaneously, making it difficult to distinguish the relative importance of water availability in relation to other factors. Thus, according to Nilsson et al. (1996) and Gemmel et al. (1996), low water potential in the soil, light level and soil temperature, as well as some unknown factor might all explain a reduction in growth of oak and beech seedlings. Löf (1999), in a study concerned with the establishment of oak, beech and Norway spruce, found soil water potential to be correlated with soil treatment, but could not relate this to growth. In another study (Löf 1999) low soil water potential was found to be the main factor affecting growth, although light and soil temperature varied.

Prediction of drought

The probability that a plant will experience drought can be predicted from plant and site properties, together with precipitation data. The plant data of importance include e.g. root distribution in relation to soil depth and transpiration rates under various climatic conditions. To define a given site the hydrological properties of the soil must be known. Such knowledge is comparatively easy to obtain. However, the soil water available may vary considerably with competition from surrounding ground vegetation and trees. The plant cover may change rather quickly making a prediction of its competitive strength difficult, since little is known of how to assess this ability. Available soil water is also dependent on precipitation, which only can be defined statistically. Since vulnerability in plants or in trees may change with age and forests may contain individuals and species of varying age a useful prediction of the impact of a drought period on the survival and growth in a forest is apparently difficult to reach.

8.4 Air pollution effects

8.4.1 Ground-level ozone

Sellén G., Karlsson P. E., Uddling J., Pleijel H. and Skärby L.

Ground-level ozone is considered to be one of the most important transboundary pollution problems in Europe and the USA. In addition, ground-level ozone is an increasing problem in other parts of the world in which traffic and industry are in a state of rapid growth. In Europe, a

cooperative research programme concerning transboundary air pollution and its chemistry has been conducted within the framework of EUROTRAC. In the assessment made, ground-level ozone is considered to be of high relevance for effects on plants, human health and materials as well as on the global climate (Borrell et al. 1997). This has led to strong efforts in the last few years, within both the scientific and the policy community. The most important actors have been UNECE and EU. Within the LRTAP-convention drawn up under UNECE, the focus has been increasingly on effect-based abatement strategies during the mid- and late 1990s. This means that abatement measures needed to be cost-effective in order to minimise the potential for negative effects. Consequently, there has been a demand for quantitative relationships between exposure and effects. In this context it has been found that the exposure index AOT40 (the Accumulated exposure Over a concentration Threshold of 40 ppb ozone) correlates well with such effects on plants as yield loss in wheat, and with growth reduction in trees (Kärenlampi and Skärby, 1996). Sweden has contributed to this development with several important pieces of information (e.g. Skärby and Karlsson 1996, Pleijel 1996). The latest multi-pollutant negotiation protocol within LRTAP was signed in Göteborg in December 1999. In addition, EU is presently finalising a new ozone directive and an adjoining ozone abatement strategy. In both cases, exposure-effect relationships are taken into account.

Level I and II mapping

Until recently, the focus of European work on critical levels for ozone has been on so-called Level I mapping. This consists of mapping of the exceedances of the critical level in different areas and countries and identification of the potential risk for ozone damage to sensitive receptors, without concern for dose- and response-modifying factors in terms of genetic differences between species and cultivars, different plant ages and those climatic factors, which are of crucial importance for plant uptake of ozone. In line with this, AOT40, the exposure index for ozone widely used in Europe for the last few years to characterise plant exposure, does not directly reflect the uptake of ozone by plants. Scientific development within this field is now at a Level II perspective, in which differences in sensitivity between plants and the consideration of other factors (light, temperature, soil and air humidity) are included. For forest trees also the transfer of data from seedlings and sapling used in experiments to mature trees in forest stands is of large importance to make accurate assessments of what is happening in the field. Only by using a Level II approach can an accurate quantification (including economic analysis) of yield losses and other effects become possible. The next generation of agreements in Europe early in the 2000s concerning abatement strategies for transboundary air pollution is likely to be based on a Level II perspective.

Assessment questions

So far, Level II work has focused on wheat, *Triticum aestivum*, and on beech, *Fagus sylvatica*. From a Nordic perspective, it is crucial to know whether forests in the far north are exposed to ozone to an extent that can cause reduction in growth. The ozone concentrations in northern Sweden are elevated, but to a lesser extent than in southern Sweden and still less than in parts of the continent. Based on the pure AOT40 concept northern ecosystems are of lesser risk than those of continental Europe. However, one of the first Level II analyses concerned with ozone (Emberson et al. 1998) indicates that the effects of ozone in the Nordic countries may be larger than expected and those in continental Europe smaller than expected. This is due to the climatic conditions favouring ozone uptake in the north. The main reasons for this are the longer daylight period during the summer in the north, as well as higher humidity in air and soil and to some extent lower temperatures.

Plant strategy in relation to ozone response

In the C-S-R system of plant strategies (Grime 1979), plants are classified in relation to responses to stress and disturbances. Grime (1979) defines stress as circumstances that limit photosynthetic production while disturbance consists of the partial or total destruction of plant biomass. On the basis of these definitions, an air pollutant such as ozone can act either as a disturbance factor or as a stress factor. When reducing net photosynthesis, ozone acts as a stress factor. When causing partial biomass destruction, (shortens the life span of the leaves) it can be considered a restricted disturbance. Since genetic characteristics affect the response of plants to stress and disturbances it is not surprising that competitors, stress-tolerators and stress-tolerant competitors respond differently to ozone stress (Selldén and Pleijel 1995).

Deciduous trees

Birch

Birch, *Betula pendula*, is sensitive to ozone and growth reductions are usually observed, especially in the roots, after a single season of exposure (Mortensen and Skre 1990, Matyssek et al. 1992, Mortensen 1993, Günthardt-Goerg et al. 1993, Pääkkönen et al. 1993, Greitner et al. 1994, Pääkkönen et al. 1995a,b, Mortensen 1996, Pääkkönen et al. 1996, Skärby et al. 1999). Furthermore, fast growing clones of birch were more sensitive to ozone than slower growing clones (Pääkkönen et al. 1993). Fast growing species, such as birch and aspen, change carbon allocation patterns rapidly in response to the ozone stress. Similar results were obtained in the Östad-birch experiment in which birch trees, *B. pendula*, were exposed to different concentrations of ozone in open-top chambers during 1997 and 1998. Elevated ozone levels significantly reduced the growth and increased the shoot-root ratio both years (Uddling et al. 2000). The ozone-induced chlorophyll loss and accelerated leaf fall in birch, *B. pendula*, *pubescens*, and *verrucosa*, and in aspen, *Populus tremuloïdes* (Mortensen and Skre 1990, Matyssek et al. 1991, 1992, Günthardt-Goerg et al. 1993, Pääkkönen et al. 1993, Greitner et al. 1994, Skärby et al. 1999) could be interpreted as accelerated senescence. Structural investigations of ozone-exposed birch clones have revealed that ozone exposure produces changes typical of senescence, such as an increase in cytoplasmic lipids, in the translucency of the mitochondrial matrix and in spherically shaped chloroplasts with protrusions (Pääkkönen et al. 1995a). Defining premature senescence as leaf death that occurs before normal life processes, such as the re-translocation of carbon and nitrogen, are terminated (Pleijel et al. 1997), it is obvious that premature senescence can occur in birch, especially under conditions of higher ozone concentrations. For example, starch was found to accumulate along the small leaf veins and in the guard cells of the leaves of ozone exposed trees, the amount increasing with increasing ozone concentration (Günthardt-Goerg et al. 1993). In the Östad-birch experiment, about 10% of the shedded leaves from the highest ozone treatment had nearly twice as high a nitrogen content as leaves from the charcoal filtered treatment (Skärby et al. 1999).

Beech

At the UN-ECE workshop in Kuopio 1996, deciduous trees were identified as the most sensitive receptors (Kärenlampi and Skärby 1996) and data from experiments with young seedlings (0-3 years) of beech, *Fagus sylvatica*, was used to set the critical level for ozone in forest trees. However, at the time the data base was very poor, since only 3 independent experiments could be used when the level was set to AOT40 of 10,000 ppb*h. Until now, the most common method used to expose trees to ozone has been the open-top chamber, most such studies having involved young trees. However, the response of young trees to ozone does not necessarily need to be the same as that of mature trees since mature trees differ from seedlings and saplings both in morphology and physiology. Results for red oak, *Quercus rubra*, indicate young trees to be less sensitive to ozone than mature trees (Samuelson and Edwards 1993, Edwards et al

1994), whereas results for giant sequoia, *Sequoiadendron giganteum*, show the opposite (Grulke and Miller 1994). On the other hand, mature and juvenile Douglas fir, *Pseudotsuga menziesii*, responded in a similar way to short-term ozone exposure (Smeulders et al. 1995). Using an epidemiological approach, Braun et al. (1999) found that stem growth of mature beech trees in Switzerland was more sensitive to ozone than young seedlings. This suggests that the present critical level for ozone (10,000 ppb*h) should be lowered to protect beech forests from substantial reductions in growth.

Conifers

It has been well established that ozone reduces net photosynthesis (Pye 1988; Darrall 1989) although a transient increase is sometimes observed in young shoots (Freer-Smith and Dobson 1989, Wallin et al. 1990, Eamus et al. 1990). At present there is no explanation for the ozone-induced stimulation in net photosynthesis. It may be an ozone-induced acceleration of the rate of development (Greitner et al. 1994, Skärby et al. 1995) or it may be an attempt of trees to compensate for the ozone-induced reduction in the net photosynthesis of the older leaves and needles (Beyers et al. 1992, Selldén and Pleijel 1995) or a combination of both. Although, photosynthesis is necessary for growth, the relationship between net photosynthesis and growth is complex, and a decrease in net photosynthesis not necessarily meaning a reduction in growth. Typical stress-tolerators depend on the rapid capture of resources during favourable conditions and the ability to store them for use later under unfavourable conditions. On the other hand, typical competitors depend less on storage than on their ability to maximise resource capture by rapid morphogenetic adjustments by reallocation of captured resources.

An ozone-induced decrease in net photosynthesis may reduce growth. However, as stored resources can be used to compensate for stress, at least for limited periods of time, species with strong stress-tolerating characteristics should be less sensitive to ozone than competitors. On the other hand, ozone exposure causing leaf or needle loss, i.e. ozone acts as a disturbance, could probably reduce the growth of stress-tolerators more than that of competitors (Selldén and Pleijel 1995). Although both competitors and stress-tolerators are adapted to habitats with a comparatively low frequency of disturbances, the low growth rates of the stress-tolerators do not allow the rapid replacement of needles or leaves. The response of trees to ozone could thus be expected to vary with species and plant strategy.

In terms of growth, conifers are less sensitive to ozone than deciduous trees are, growth reductions rarely being observed after a single season of exposure. For example, in white pine, *Pinus strobus*, ozone exposure for 4 months was found to reduce net photosynthesis but not growth (Reich et al. 1987). In red spruce, *Picea rubens*, growth was not affected until after the end of the second exposure season (Amundson et al. 1991) and in Norway spruce exposed for two seasons in open-top chambers growth was not reduced at all (Nast et al. 1993). Similarly, Braun and Flückiger (1995) and Lucas and Diggle (1997) observed no effects on stem volume increment or on growth of Norway spruce, respectively, after 3 years of ozone exposure. On the other hand, Payer et al. (1990) found 2 years of ozone exposure to reduce the stem diameter increment in 2 out of 5 clones of Norway spruce. Polle et al. (1996) and Lippert et al. (1997) also observed ozone-induced growth reduction in Norway spruce. It could be suggested that Norway spruce is not very sensitive to ozone, but the contradicting results may also be explained by a high variability among the plants and short exposure periods, in combination with the low numbers of replicates.

In the Östad-spruce project a total of 828 Norway spruce saplings from one clone were exposed to ozone, either alone or in combination with low phosphorous supply or with drought, in 42 open-top chambers during 4 growing seasons (1992-1995). The results showed that moderately elevated levels of ozone reduced the relative growth rate by approximately 2% (Skärby et al. 1999). The average annual AOT40 value for the ozone treatment during the four years was 20 000 ppb*h, which is twice the present critical level. To illustrate the possible consequences of a 1% reduction in the relative growth rate, corresponding to 10 000 ppb*h, simple model cal-

| | Plot 1 | Plot 2 | Plot 3 | Plot 4 | Plot 5 |
|---|-----------|-----------|--------|--------|--------|
| soil | fine sand | fine sand | sand | till | till |
| % spruce | 100 | 87 | 87 | 100 | 100 |
| age | 26 | 32 | 32 | 32 | 31 |
| stems ha ⁻¹ | 1900 | 1650 | 1650 | 1200 | 2100 |
| m ³ sk ha ⁻¹ | 105 | 135 | 135 | 150 | 140 |
| thinned | no | yes | yes | yes | no |
| m ³ sk ha ⁻¹ yr ⁻¹ * | 9.5 | 7.5 | 11.0 | 11.8 | 9.3 |
| m ² ha ⁻¹ yr ⁻¹ * | 1.0 | 0.6 | 0.9 | 1.0 | 0.8 |

Table 8.6: *Site characteristics of the spruce plots selected for the ozone/growth study. The data are from 1990, except the parameters marked by *, which show growth per year 1992-96.*

culations was made. They showed that if ozone only affected the growth of the young plant, the stem volume of a spruce tree of "harvestable age" would be reduced by about 3%. The reduction would be 8% if ozone affected the trees throughout the phase of intensive growth (0-20 years). If ozone depressed growth throughout the entire life cycle, the stem volume would be reduced by about 11%, as compared with an environment with pre-industrial ozone levels (Skärby et al. 1999). In an attempt to assess the impact of ozone on mature Norway spruce trees in Sweden, use was made of an epidemiological approach, the influence of ozone on the stem basal area increment being tested on 24 Norway spruce trees distributed over five investigation plots at Asa Research Park similar in stand conditions and in growth patterns, Tab.8.6. Analysis of the correlations between basal area increment and environmental parameters, determined on a weekly basis, showed there to be a statistically significant negative correlation between the basal area increment and both the average daylight ozone concentrations (ozoneavg, as well as with AOT40 per day. This indicated ozone to have an impact on the weekly stem basal area increment. However, the magnitude of the effect, as well as the long-term significance, remain to be established. Other parameters having significant impact were day of the year, precipitation (24h average, mm h⁻¹), air temperature (24h average, C), soil water potential at 10 cm depth (24h average, MPa) and irradiance (24h average, W m⁻²).

During 1993-1999, stem circumferences on five trees per plot were measured weekly from the beginning of May until the end of August. In 1995 there was a lack of ozone data for May and the growth measurements started in June. Data on air temperature, relative humidity, global radiation, precipitation and ozone concentration were provided by the meteorological station at Asa Research Park on an hourly basis. In addition, in each stand the soil water potential was measured weekly at 10 and 40 cm depth with 5 gypsum blocks per depth. Stem growth was calculated as the basal area increment (BAI), assuming both linear (8.10) and logarithmic (8.11) growth:

$$\text{linear stem growth} = \frac{BA_2 - BA_1}{t_2 - t_1} \quad (8.10)$$

$$\text{logarithmic stem growth} = \frac{\ln(BA_2) - \ln(BA_1)}{t_2 - t_1} \quad (8.11)$$

where BA is basal area. The statistical methods used were similar to those employed in epidemiological studies in human medicine. The correlation between stem area increment and environmental parameters were analysed on a weekly basis using stepwise multiple regression analysis, the dependent parameters being log (stem growth) and lin stem growth and the independent parameters being average ozone, ozon AOT40, year, yearday, soilP10cm, temperature, irradiance and precipitation. VPD correlated with all the independent variables except for

the yearday and ozonavg thus not being included in the final statistical analysis. The models explained 22-24% of the total variance. All the parameters included in the model were significant, $p < 0.0001$, except for irradiance which was significant at $p = 0.044$ (8.12) and $p = 0.078$ (8.13). However, further analyses using multivariate statistics will be carried out to allow the VPD to be included in the calculation. We have:

$$BAI = 16.3 + 29.03 \cdot P + 1.16 \cdot T - 0.109 \cdot \text{yearday} - 0.21 \cdot [O_{3\text{avg}}] + 2.002 \cdot \theta - 0.0083 \cdot F \quad (8.12)$$

$$BAI = 6.79 + 30.93 \cdot P + 1.064 \cdot T - 0.0876 \cdot \text{yearday} - 0.0123 \cdot AOT40 + 2.203 \cdot \theta - 0.0073 \cdot F \quad (8.13)$$

BAI being basal area increment, P precipitation, T temperature, F irradiance and theta WP10. Statistical analysis of the correlation between the basal area increment and the environmental parameters on a yearly basis was hampered by the limited amount of data available.

Diagnosis of ozone injury

The 1997 Executive Report on forest conditions in Europe stated that the steady increase in defoliation over large areas of Europe to be a matter of concern. Natural factors such as drought and anthropogenic factors such as air pollutants can be considered to be major factors for forest damage, but the difficulties at the current state of knowledge in establishing cause-relationships in multi-stress situations, are acknowledged. However, the same report states further that "an implementation of in-depth studies of cause-effect relationships on a smaller number of plots might be considered by ICP-Forest in the future". A lack of good diagnostic methods for a factor-specific assessment of the effects of ozone, as well as other relevant air pollutants, was noted by the working sub-group on Ambient Air Quality (Deposition Expert Panel of ICP-Forest) in Essen in March, 1999. Microscopic morphometry, at both the transmission electron microscope and the light microscope level, has shown that both needle and leaf structure exhibit highly specific changes in response to different stress factors such as drought, ozone, acid rain, sulphur dioxide and nitrogen dioxide. Regardless of tree age, normal chloroplasts are lens-shaped, the stroma being light and the amount of plastoglobuli usually being small (Wallis et al. 1973, Soikkeli 1978, Sutinen 1987a, Meyberg et al. 1988, Holopainen and Nygren 1989). Whether trees are exposed in a laboratory exposure system or in the field, by use of open-top chambers or of open-release systems, ozone-induced changes are always observed first in the chloroplasts. The areas of the chloroplasts decrease, the density and the granulation of the stroma increase and the thylakoid membranes become difficult to resolve. Similar changes have been observed in mature Norway spruce and Scots pine in the field in Finland (Sutinen et al. 1995, Sutinen et al. 1998) and in Norway spruce in central Europe (Sutinen 1987b). Furthermore, ozone-induced structural changes have also been observed in 1-year-old needles of mature Norway spruce from Asa Research Park.

Conclusions

The growth of young birch trees is more sensitive to ozone than that of young Norway spruce trees. The Östad- spruce project showed the exposure of Norway spruce to 80,000 ppb*h to result in a 5% reduction in the biomass whereas the exposure of birch to 40,000 ppb*h resulted in a 20% decrease. Microscopic morphometry can be used to assess to what extent mature Norway spruce trees are affected by ozone. Epidemiological studies of the short-term stem basal area increment indicated ambient ozone concentrations to affect the growth of mature Norway spruce trees. The Critical Level II approach, taking into account e.g. species and environmental factors affecting the uptake of ozone, is necessary for any accurate assessment of the impact of ozone on forests, as well as for economic consequences. The European scientific community is now moving towards the Critical Level II approach since the next generation of agreements in Europe early in the 2000s concerning abatement strategies for transboundary air pollution is likely to be based on the Level II perspective. Also, one of the first Level II analyses

related to ozone indicates that the effects of ozone may be larger than expected in the Nordic countries and smaller than expected in continental Europe.

8.4.2 Acid deposition and soil acidity

Ingrid Stjernquist, Harald Sverdrup and Torkel Welander

The discussion about forest health in relation to soil acidification has identified three principal ways in which acid soils can reduce tree vitality

- An increase in Al^{3+} , caused by a decrease in pH below 4.5, negatively affects nutrient uptake and growth
- Soil acidification causes leaching of base cations from the root zone, reducing the available nutrient pool.
- Soil acidification has direct negative effects on growth and nutrient uptake

During the second half of the 20th century, the acidification of southern Swedish forests soils has accelerated, in the last 50 years pH in the upper soil layer having declined on an average by 0.8 pH units. Both podzols and cambisols have been affected. Forest management and air pollution deposition are the main agents involved (e.g. Hallbäcken and Tamm 1986a, Falkengren-Grerup 1987). In southwestern Sweden, with its high deposition of air pollutants, the weathering capacity of the soil minerals are not high enough to compensate for the acid input (Karlton 1994). Hallbäcken and Tamm (1986b) in evaluating a forest in southwest Sweden, concluded that the increase in productivity over time is only responsible for a minor part of the increase in soil acidity. However, the change in utilising forests as a resource that supports a farm as an economic unit to an intensive silviculture dominated by conifers probably contributes to a high degree to soil acidification, due to the increase in nutrient uptake from the forest soils.

In an investigation on the pH of streams in 48 small catchment areas differing in land use and in species composition concluded that an increased amount of spruce in a landscape significantly decreases the pH (Sigvardsson-Lööf 2000). For a detailed historical review of soil acidification in southern Sweden and of the resulting chemical changes, see Nilsson and Tyler (1995). Despite the acidification caused by growth, spruce is the most efficient species for capturing air pollutants through the filtering effect of its crowns. The filtering capacity depends on the leaf area, the leaf characteristics and the structure of the tree. In addition, evergreen conifers generally are able to capture pollutants during a longer time period than deciduous trees. The difference between species also increase with the load of pollutants (Liljelund et al. 1986, Balsberg-Påhlsson and Bergkvist 1995). Comparisons between species show that conifers are more acidifying than birch, the difference being 0.3 - 0.7 pH units in the mineral soil. A conifer stand containing 50% birch increases the pH in the humus layer by 0.2-0.3 units as compared with a spruce monoculture (Liljelund et al. 1986). According to Bergkvist and Folkesson (1995), beech affects the soil pH less than spruce but more than birch. Moreover, the influence of deciduous tree species on pH in the upper mineral soil layer, 0-5cm, depends on species. The influence increases in the order *Tilia - Acer - Quercus/Carpinus - Fagus*, *Tilia* being the least acidifying species. The effect is independent of soil type. Compared to oak, the soil pH below beech was 0.13 units lower and that below *Acer* and *Tilia* was 0.15 and 0.32 units higher, respectively (Nordén 1994). During the last decades the soil profile, even the layers beneath the root zone, shows an increase in acidification. This cannot be explained by root activity alone. For many decades, the deposition of acidifying sulphur and nitrogen compounds has been an important stress factor (Hallbäcken and Tamm 1986a). Danish investigations show that despite reduction in sulphur deposition during the period of 1985 to 1993, soil acidification is still increasing (Hovmand and Kemp 1996). In total, the acidifying factors have decreased the pH in Swedish soils below 5 for 2 mill ha of forestland south of the region of Wärrmland

between 1983/85 and 1993/95. Soil acidification influences not only the soil characteristics but also forest biodiversity and water quality.

Changes in the nutrient pool

Falkengren-Grerup and Tyler (1992) have estimated that in southern Sweden during the period of 1950 - 1988 about 50% of the soil content of essential nutrients (K, Ca, Mg) in the C-horizon disappeared, the pool of aluminium during this period becoming twice as high. During the 1980s, when the total deposition of acidifying compounds was high and constant, the top layer (0-5cm) of beech forest soils lost a significant amount of K, Ca and Mg, the largest changes being in the least acid soils (Falkengren-Grerup and Tyler 1991). A risk assessment of the southernmost part of Sweden indicated that 65% of the investigated sites had a deficit of Ca, Mg or K (Barkman and Sverdrup 1996). Results for permanent coniferous forest plots in the same area showed that in the late 1980s, the base saturation percentage in the mineral soil, 20-30 cm, was only 19-26% and that the content of P and K were affected most (Nihlgård 1990). A new investigation in 1999 confirmed those negative trends (Jönsson et al. 2001). That year, the base saturation on 80% of the plots was below 10%, K and Mg being the nutrients most negatively affected. The pH of water on 70% of the plots was below 4.2 and on 84% of the plots the aluminium value had increased to above the recommended maximum level of 100µg/g. The influence of various species on the nutrient pool may well be significant, in particular if conifers are grown instead of deciduous trees. In an area of beech forest with a continuity since the beginning of the eighteenth century, small 50 year-old spruce stands exhibited a 25% reduction in the K and Mg levels in the mineral soil as compared with the earlier beech stand.

Base cations and aluminium

When the pH decreases to below 4.5, the Al^{3+} concentration in the soil increases exponentially. A high aluminium content in the root zone causes root growth reduction and damage. Although up to now the field levels of aluminium in the upper soil layers have generally not reached concentrations considered toxic to plants (e.g. Thornton et al. 1989, van Praag and Weissen 1985). However, the soil analyses commonly employed may undervalue the situation in the near root zone (Göttlein et al. 1999).

The aluminium concentrations in southern Swedish soils are in the range of 0.01 to 0.7mM, depending on the site and the tree species (Bergkvist 1987, Bergkvist and Folkesson 1992, 1995, Ljungström and Stjernquist 1993). The ability of the aluminium ion to complex binding with fulvic or organic acids diminishes the amount of toxic Al in the soil, which might reduce the negative effects on root growth and on nutrient uptake (Foy et al. 1978, Asp and Berggren 1990).

Al^{3+} inhibits nutrient uptake through competing with base cations, especially calcium and magnesium, at the root surface (Ericsson et al. 1995b). Disturbances in nutrient uptake occur at lower Al^{3+} concentrations than are needed for the direct inhibition of root growth (Godbold et al. 1988). The fact that aluminium interferes with root uptake of essential nutrients, especially Ca, is the basis for the formulation of the growth/aluminium relationship. Tree growth decreases when the ratio of Ca or base cations to aluminium, the BC/Al ratio, are close to 1 (Rost-Siebert 1984, Sverdrup and Warfvinge 1995). Root length is reduced to up to 40% when the Ca/Al ratio declines from 10 to 1 (Neitzke 1989) and a further decline in root length is found at a still lower Ca/Al ratio, 0.5 (Rost-Siebert 1984). However, there seems to be between species differences in the reaction to aluminium as reviewed in Tab.8.7 and Tab.8.8.

Aluminium in combination with nitrogen further inhibits calcium uptake. High concentrations of ammonium or of nitrate, 5.6 mM, in combination with 0.1mM Al^{3+} was found to decrease the Ca uptake by 50%, as compared with a situation with an excess of nitrogen only (Bengtsson 1992, Bengtsson et al. 1994). In contrast to other nutrients, the phosphorous and potassium uptake in beech seedlings grown in a nutrient solution was stimulated by aluminium

| Species | mM Al ³⁺ | Effect | Reference |
|---------------|---------------------|-----------------|--|
| Norway spruce | 0.04- 1.2 | seedling growth | Rost-Siebert 1983, Tischner et al. 1983 |
| | 0.8- 1.2 | root length | Godbold et al. 1988 |
| | 0.3 | seedling growth | Göransson & Eldhuset 1991, 1995 |
| Scots pine | 0.1 | root/shoot | Godbold & Jentschke 1998 |
| | 0.04-1.2 | seedling growth | Rost-Siebert 1983, Tischner et al. 1983 |
| | 0.8-1.2 | root length | Godbold et al. 1988 |
| Birch | 6 | seedling growth | Göransson & Eldhuset 1991, Ericsson et al. 1995b |
| | 3 | seedling growth | Göransson & Eldhuset 1991, Ericsson et al. 1995b |
| Beech | 0.07- 0.11 | seedling growth | Ulrich et al. 1980 |
| | 0.1 | seedling growth | Bengtsson et al. 1988 |
| | 0.5 | shoot growth | Kelly et al. 1990, Thornton et al. 1989 |
| | 1 | shoot growth | Balsberg-Påhlsson 1990 |

Table 8.7: *Aluminium concentrations that hamper seedling growth in Norway spruce, Scots pine, birch and beech.*

in low concentrations, 0.1mM (Bengtsson et al. 1988, 1994). This positive effect has also been found for spruce (von Jorns and Hecht-Buchholz 1985). Balsberg-Påhlsson (1990) found the uptake of phosphorus in beech seedlings to be stimulated by Al³⁺ in concentrations relevant for soil solutions in southern Sweden, but to be reduced at higher Al³⁺ concentrations. This indicates that under field conditions the aluminium - nutrient relationship is not so straightforward as many laboratory experiments with high concentrations of aluminium indicate. The long-term reaction of trees to aluminium concentrations that hamper Ca and Mg uptake but stimulate P and/or K uptake should be investigated further in relation to the nutrient imbalance found in the Swedish soils. Besides the direct effects it has on growth and nutrient uptake, an increased Al³⁺ concentration in the soil also cause indirect effects. The carbon allocation in tree seedlings is affected by aluminium. The carbohydrate and phenol content in beech seedling roots increases at levels of 1 and 0.5 mM Al³⁺ respectively, the starch concentration in the shoots being enhanced at 1mM Al³⁺. This effect can be interpreted as mainly being due to disturbances in carbohydrate metabolism (Balsberg-Påhlsson 1990). A similar carbon allocation response has been found for pine (Arovaara and Ilvesniemi 1986). If low pH and high Al³⁺ concentrations in the soil are counteracted by liming or vitality fertilization, the balance between Ca and Mg may change, resulting in a shift in the mutual competition between the cations. For example, an Mg concentration that is non-limiting for growth may be inadequate in the presence of an excess of Ca (Mengel and Kirkby 1987, Ericsson et al. 1995).

To summarize, aluminium in concentrations representative for southern Swedish forest soils appear to be unable to affect root growth directly. However, through interference with nutrient uptake, aluminium suppresses the uptake of Ca, Mg as well as of Zn, K and P in soils with high amounts of Al³⁺. Reduced Ca and Mg uptake are considered to be the main effect of aluminium on tree vitality (van Praag et al. 1991). The N deposition found in southern Sweden probably increase the aluminium effect.

H⁺ toxicity

The nutrient uptake in the spruce roots is not affected by the H⁺ ions if the roots can adjust the pH up to neutral values. Kuhn et al. (1995) have identified a threshold value at pH 2.5. Under more acid conditions, no adjustment is possible, the K uptake decreasing in particular.

Field experiments with beech seedlings growing in acid cleyey till (dystric cambisol, pHKCl

| Species | mM Al ³⁺ | Nutrient | Reference |
|------------------|---------------------|------------|---------------------------|
| Norway spruce | 0.1 | Ca, Mg | Godbold & Jentschke 1998 |
| Norway spruce | 0.2 | Ca, Mg | Göransson & Eldhuset 1991 |
| + ectomycorrhiza | 2 | Ca, Mg, K | Kuhn et al. 1995 |
| Scots pine | 1 | Ca, Mg | Göransson & Eldhuset 1991 |
| + ectomycorrhiza | 3 | Ca, Mg | Göransson & Eldhuset 1987 |
| Birch | 1 | Ca, Mg | Göransson & Eldhuset 1991 |
| Beech | 0.1 | Ca, Mg, Zn | Balsberg-Påhlsson 1990 |
| | 0.5 | Ca, Mg, P | Thornton et al. 1989 |
| | 0.1/1 | Ca, Mg/P | Bengtsson et al. 1988 |

Table 8.8: Aluminium concentrations that hamper nutrient uptake in Norway spruce, Scots pine, birch and beech.

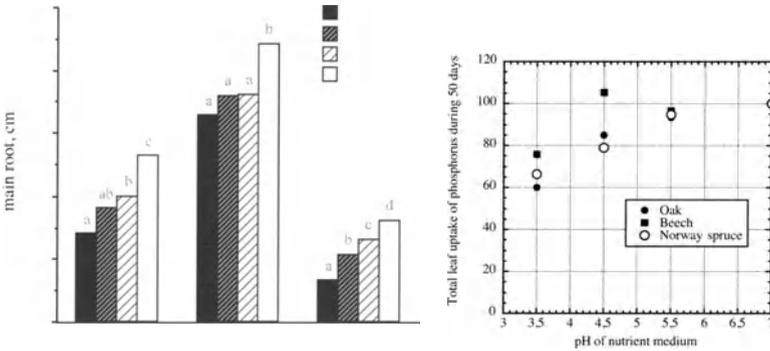


Figure 8.7: To the left: Root length of beech, oak and Norway spruce seedlings grown in nutrient solution with pH 3.5, 4.5, 5.5 and 7 (Stjernquist and Welander 2002). To the right: Response in P uptake to the leaves as a function of pH.

3.32) and podzol (acid haplic podzol, pHKCl 2.90) in southern Sweden indicate H⁺ toxicity and H⁺ related factors to have a strong effect in reducing plant growth and root development (Ljungström and Stjernquist 1993). This can probably be explained by the imbalance in nutrient uptake. The effect of pH on the root architecture differs between species. Thus, the root length of beech and spruce, but not of oak, decreases significantly when pH becomes lower, Fig. 8.7. The data shown in Fig. 8.7 can readily be fitted to an isotherm such as those developed for BC/Al ratios, and be used as a response function in models

$$f(pH) = \frac{1}{1 + k_{pH} \cdot [H^+]^n} \tag{8.14}$$

where k_{pH} and n are determined empirically. McLaughlin and Wimmer (1999) found that a decrease in root length hampers the nutrient capacity more than a decrease in root weight does. Like aluminium, the H⁺ ion hampers the uptake of Ca and Mg (Godbold and Jentschke 1998, McLaughlin and Wimmer 1999). The effect seems to be related to tree species. Rost-Siebert (1984) reported beech to be more sensitive than Norway spruce to high acidity

per se, at least 8 times as high a Ca concentration being needed to counteract the H^+ toxicity effects in beech. Experiments under controlled climate conditions, using nutrient solutions at a pH of 7.0 to 3.5 clearly showed a difference in nutrient uptake capacity between oak, beech and Norway spruce, Fig. ?? . For Norway spruce, pH values below 4.5 hamper the uptake of Ca, P and K, whereas for beech the Ca, Mg and K uptake decreases. Oak appears to be less sensitive to an excess of H^+ , the Ca and Mg uptake decreasing only at the lowest pH (Stjernquist and Welander 2002). Under field conditions, oak seedlings grown in very acid soils, pH_{KCl} 3 show the expected decrease in Ca and Mg uptake. In beech, the P uptake is affected at lower pH but with a stronger response than the other species. In contrast to oak, beech seedlings have a low percentage of fine roots, indicating that additional soil factors may act together with low pH (Ljungström and Stjernquist 1995, Sonesson 1994).

To summarize, H^+ affects the uptake of Ca, Mg, K and P negatively and the effect can be modelled by use of simple response functions. However, there are differences in response between species, these being expressed in the functions as different values for the parameters k_{pH} and n . For beech and oak, the uptake of Ca and Mg is affected most, and for Norway spruce the uptake of Ca, P and K.

Tree species with large seeds, such as oak and beech, have a large pool of nutrients available for early seedling growth. The nutrient content of the seeds is dependent on soil characteristics. If air pollutant deposition or forest management decreases the available pool of essential nutrients, the capacity for seedling growth and survival may be negatively affected. In a study in southern Sweden, the quality of beech seeds in relation to soil acidification was measured. The result indicates that both seed weight and nutrient content, especially the total P and K content, were negatively affected by a decrease in soil pH (Stjernquist 2002b).

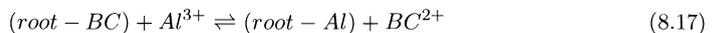
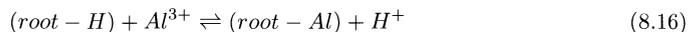
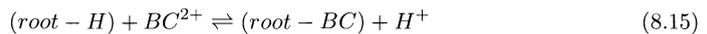
Conclusion

For Swedish soils the effects of an increased acidification and in the concentration of aluminium are:

- Soil acidity affects tree vitality through changes in nutrient uptake capacity in particular. Generally, the uptake of Ca, Mg and K appears to be most disturbed.
- There appears to be a difference between species regarding which of the nutrients are most affected by Al^{3+} and by H^+ , respectively. If the two stress factors act together, beech and oak are particularly sensitive to a combination of aluminium and H^+ .

The valence unspecific mechanism

For a valence unspecific reaction the ion exchange matrix is indifferent to the valence of the adsorbing ions, the matrix behaving as an infinite continuum of receptor sites. This implies that whenever a base cation is absorbed, one H^+ or Al is released, the difference in change being adjusted for by some at present unknown process:



the valence unspecific selectivity coefficients for root exchange becoming

$$K_{H/BC} = \frac{X_{BC}}{X_H} \cdot \frac{[H^+]}{[BC^{2+}]} \quad (8.18)$$

$$K_{H/Al} = \frac{X_{Al}}{X_H} \cdot \frac{[H^+]}{[Al^{3+}]} \quad (8.19)$$

$$K_{BC/Al} = \frac{X_{Al}}{X_{BC}} \cdot \frac{[BC^{2+}]}{[Al^{3+}]} \quad (8.20)$$

where X_i is the exchanged fraction of substance i , the parameters in brackets being concentrations and K the selectivity coefficients. The equation for divalent base cations uptake by trees is obtained by combining the growth rate equation with the expression for X_{BC} and the expressions for the adsorbed fraction of the base cation, substituting for the selectivity coefficients, depending on reaction type. This may be used to solve for X_H and X_{Al} , assuming BC, Al and H to be the dominating ions on the surface such that:

$$X_{BC} = 1 - X_{Al} - X_H \quad (8.21)$$

To obtain an expression for the base saturation in the roots by rearranging the selectivity expressions, to expressions for exchangeable H and Al at the root as function of the base saturation X_{BC}

$$X_H = X_{BC} \cdot \frac{1}{K_{H/BC}} \cdot \frac{[H^+]}{[BC^{2+}]} \quad (8.22)$$

$$X_{Al} = X_{BC} \cdot K_{H/Al} \cdot \frac{[Al^{3+}]}{[BC^{2+}]} \quad (8.23)$$

These are then filled in Eq. 8.21

$$X_{BC} = 1 - X_{BC} \cdot K_{H/Al} \cdot \frac{[Al^{3+}]}{[BC^{2+}]} - X_{BC} \cdot \frac{1}{K_{H/BC}} \cdot \frac{[H^+]}{[BC^{2+}]} \quad (8.24)$$

The response function $f(BC/Al)$ is obtained by setting the fraction of base cations at the root surface to

$$f(BC/Al) = \frac{U}{U_{max}} = X_{BC} \quad (8.25)$$

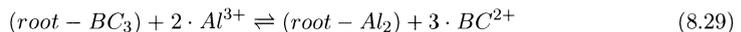
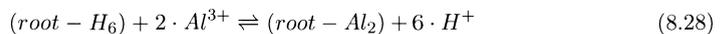
By changing the notation of the coefficients, the response function $f(BC/Al)$ can be given by

$$f(BC/Al) = \frac{[BC^{2+}]}{[BC^{2+}] + K_{Al} \cdot [Al^{3+}] + K_H \cdot [H^+]} \quad (8.26)$$

As can be seen, this is the Michaelis-Menten equation for the uptake of base cations, the difference being that the saturation coefficient is dependent on competing ions for uptake positions, such as H and Al. Theoretically, any ion not useful for uptake can have such an effect. The valence unspecific mechanism implies there to be no BC-antagonism toward Al beyond the first order relation, and there to be only one isotherm in terms of BC/Al ratio, regardless of Ca or Mg concentration.

The Vanselow mechanism

For a variant of the Vanselow type of reaction, the root ion exchange matrix is viewed as a polydentate valence specific substrate. During the reaction, a rearrangement of the solid phase must occur in order for three BC^{2+} -ions or two Al^{3+} -ions to be linked to a hexa-valent binding site by double bonds (Warfvinge, 1988). The reaction stoichiometry suggested is



The Vanselow selectivity coefficients become by application of the law of mass action, in view of the activity of a phase in an ideal mixed crystal depending on the molar fraction of the species:

$$K_{H/BC} = \frac{2 \cdot X_{BC}}{X_H} \cdot \frac{[H^+]^6}{[BC^{2+}]^3} \quad (8.30)$$

$$K_{H/Al} = \frac{3 \cdot X_{Al}}{X_H} \cdot \frac{[H^+]^6}{[Al^{3+}]^2} \quad (8.31)$$

$$K_{BC/Al} = \frac{3 \cdot X_{Al}}{2 \cdot X_{BC}} \cdot \frac{[BC^{2+}]^3}{[Al^{3+}]^2} \quad (8.32)$$

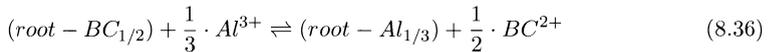
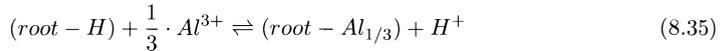
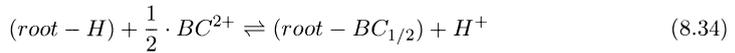
The Vanselow mechanism implies that the root surface receptors have a fixed valence that is hexagonal orientated. This suggests the coordination of water Ca-, Mg- and Al-hydrate-complexes to receptor sites. For a Vanselow root ion exchange mechanism reaction, the growth inhibition expression can be derived by the same rearrangement of the terms mathematically as was demonstrated for the unspecific response mechanism. The response expression is

$$f(BC/Al) = \frac{[BC^{2+}]^3}{[BC^{2+}]^3 + K_{Al} \cdot [Al]^2 + K_H \cdot [H]^6} \quad (8.33)$$

The Vanselow mechanism implies there to be a strong antagonistic effect of Ca, Mg and K toward Al, as well as a strong effect of the BC/Al-ratio on growth, since it is second order with respect to this ratio, implying too that plants with this type of reaction should react strongly in their uptake of base cations to changes in the base cation concentration. The change is one which has a relatively stronger effect on uptake in relation to plants with an unspecific response. This implies there to be several different isotherms for a given BC/Al-ratio depending on the BC concentration in the soil solution.

The Gapon mechanism

The Gapon ion exchange mechanism implies exchange of ions based on charge expressed as equivalents. Each BC^{2+} -ion and Al^{3+} -ion is bound to a binding site by a single bond which maintain their valence at -1. The stoichiometry of the ion exchange reactions occurring at the root between the surface, H^+ , BC^{2+} and Al^{3+} are for a Gapon reaction:



For the Gapon reaction the selectivity coefficients, after applying the law of mass action using single-bonded H^+ , BC^{2+} and Al^{3+} and charge fractions on the ion exchange matrix, become

$$K_{H/BC} = \frac{X_{BC}}{X_H} \cdot \frac{[H^+]}{[BC^{2+}]^{1/2}} \quad (8.37)$$

$$K_{H/Al} = \frac{X_{Al}}{X_H} \cdot \frac{[H^+]}{[Al^{3+}]^{1/3}} \quad (8.38)$$

$$K_{BC/Al} = \frac{X_{Al}}{X_{BC}} \cdot \frac{[BC^{2+}]^{1/2}}{[Al^{3+}]^{1/3}} \quad (8.39)$$

For a Gapon root reaction mechanism, the uptake inhibition expression, after use of the gibbsite expression is

$$f(BC/Al) = \frac{[BC^{2+}]^{1/2}}{[BC^{2+}]^{1/2} + K_G \cdot ([Al] + p \cdot [H^+])^{1/3}} \quad (8.40)$$

where p is the ratio between the Gibbsite coefficient and the H/BC ion exchange selectivity coefficient at the root surface. The Gapon mechanism, which may be the most common mechanism for cation ion exchange in dead organic matter in the soil (Warfvinge 1988), imply charge surface balance and charge exchange. This appears to seldom occur on parts of living plants.

The Gaines-Thomas mechanism

No ion exchange analogy can be investigated without trying out the Gaines-Thomas equation, which has been employed in many soil chemistry models. The Gaines-Thomas reaction mechanism leads to an expression intermediate between the unspecific response expression and the Vanselow expression, with the exception that the reaction sites maintain their valence at -1. This implies BC^{2+} ions being bound to two single bond sites by single bonds. This isotherm differs in practice very little from the unspecific expression, but retains a small antagonistic effect of Ca and Mg towards Al in addition to the 1:1 built into the BC/Al ratio.

Discussion of the mechanisms

The general equations for the damage functions can all be expressed in terms of the BC/Al-ratio in order to highlight any additional antagonistic effects. This can also be seen as a purely empirical formula based on a form such as

$$f(BC/Al) = \frac{[BC^{2+}]^n}{[BC^{2+}]^n + K_{Exp} \cdot ([Al^{3+}] + p \cdot [H^+])^m} \quad (8.41)$$

which is equivalent to

$$f(BC/Al) = \frac{[BC^{2+}]^{n-m} \cdot (BC/(Al + p \cdot H))^m}{[BC^{2+}]^{n-m} \cdot (BC/(Al + p \cdot H))^m + K_{Exp}} \quad (8.42)$$

where n and m are exponents and K_{Exp} is a coefficient that needs to be determined experimentally. For analysis of the data, $BC/(Al+p \cdot H)$ can be regarded as the parameters one should use to plot response versus soil acidity. This implies there to be several different isotherms for the same BC/Al-ratio, depending on BC concentration in the soil solution when n and m are different. An important consequence of using the empirical expression is that it allows valid response curves to be determined, even when the actual mechanism of response at the molecular level is completely unknown. Al can be removed from the expression for conditions where the soil solution concentration of Al is insignificant. The relationship between the BC/Al-limit and the BC/H-limit can be determined at the point where the response functions have the same value. From this, the relation between the BC/H-limit and the BC/Al-limit can be derived.

$$\left(\frac{BC}{Al}\right)_{limit} = \frac{1}{p} \cdot \left(\frac{BC}{H}\right)_{limit} \quad (8.43)$$

For the "unspecific" response type, $p=1$, the BC/H-limit is equal to the BC/Al-limit. For the Vanselow response $p=3$, the BC/H-limit is equal to the 3 times the BC/Al-limit. For the Gapon response, $p=1$ (Jönsson et al. 1995).

Turning back to the theoretical derivation of the expressions at the beginning of the book, recall that Nye Tinker (1977) suggested the generic Michaelis-Menten expression for uptake. In our treatment of the uptake process, we arrived at something similar, starting from the first order uptake expression for the base cations, assuming the amount adsorbed on the root surface

to be the determining parameter for uptake. Considering the competition between Al, H and BC at the surface for uptake acceptor sites, one can show that the uptake expression is:

$$r_{up} = k_{up} \cdot \frac{[BC^{2+}]^n}{[BC^{2+}]^n + K_{Exp} \cdot ([Al^{2+}] + p \cdot [H^+]^m)} \quad (8.44)$$

This is a Michaelis-Menten rate expression, based on the concentration of substrate in the soil solution. The half-rate saturation coefficient of the Michaelis-Menten expression in this approach is affected by the presence of Al and H ions in the soil solution. The difference as compared with the earlier Michaelis-Menten expression suggested by Nye and Tinker (1977) is that, whereas the old expression ignores soil chemical conditions except BC availability, the present expression also includes the effect of Al and H in the soil solution. The response function derived here has no effect if the other nutrients are more limiting than the base cations when constricted by Al and H effects. It is thus the maximum uptake capacity that is constricted by these functions. The general equations for the damage functions can all be expressed in terms of the BC/Al-ratio in order to reveal any additional antagonistic effects. In summary, the following generalized expression can be used:

$$f(BC/Al) = \frac{[BC^{2+}]^n}{[BC^{2+}]^n + K([Al^{3+}] + p \cdot [H^+]^m)} \quad (8.45)$$

where n and m are exponents and K_{Exp} is a coefficient to be determined experimentally. From the theoretical considerations made above, the following values can be suggested. These values have been confirmed for grasses using the studies of Anderson and Brunett (1993) and later also for fungi and bacteria in a laboratory culture (Jönsson et al. 1994). We have after Sverdrup and Warfvinge (1993):

| Parameter | n | m | p | Applicable to |
|---------------|-----|-----|---|---------------------------------|
| Unspecific | 1 | 1 | 1 | Spruce, Pine, Grasses, Flowers |
| Vanselow | 2 | 3 | 3 | Deciduous trees, Crops, Flowers |
| Gapon | 1/2 | 1/3 | 1 | Willow, Coffee |
| Gaines-Thomas | 1.5 | 1 | 1 | Spruce, Pine, Grasses |

Experimental data is required to determine which of the response types apply to a certain damage function. Experimental data seem to indicate that the uptake of N and P is also restricted by the Al response. Other surface reaction stoichiometries than those shown above were tested, but none except those given above fitted the data reasonably well.

8.4.3 Experimental results

When called for, the data used in this study was normalized by use of control experiments. Fig. 8.9 show the results of laboratory assays for Norway spruce (*Picea abies*), excluding the data of Arovaara and Ilvesniemi (1990). The spread of the data reflects the natural variation in sensitivity of the plants within a given species. Such data have been much questioned, since current methods have not shown the capacity to detect such effects under field conditions with any degree of accuracy. Regional data are however forthcoming that appear to confirm the existence of this kind of relationships in central Europe. The distribution of observed points around the BC/Al-response function is a normal distribution, with a standard deviation of +/-15%, $r^2=0.67$. The plot shows that one cannot expect to set sharp limits but must always work with a degree of uncertainty. This is not a problem in large samples if the uncertainty, as in this case, is symmetrical. Then stable averages can be made and the average uncertainty may be very small. All the spruce species investigated follow the unspecific response consistently (Sverdrup and Warfvinge, 1993). The data of Arovaara and Ilvesniemi (1990) and Ilvesniemi (1992) on seedlings younger

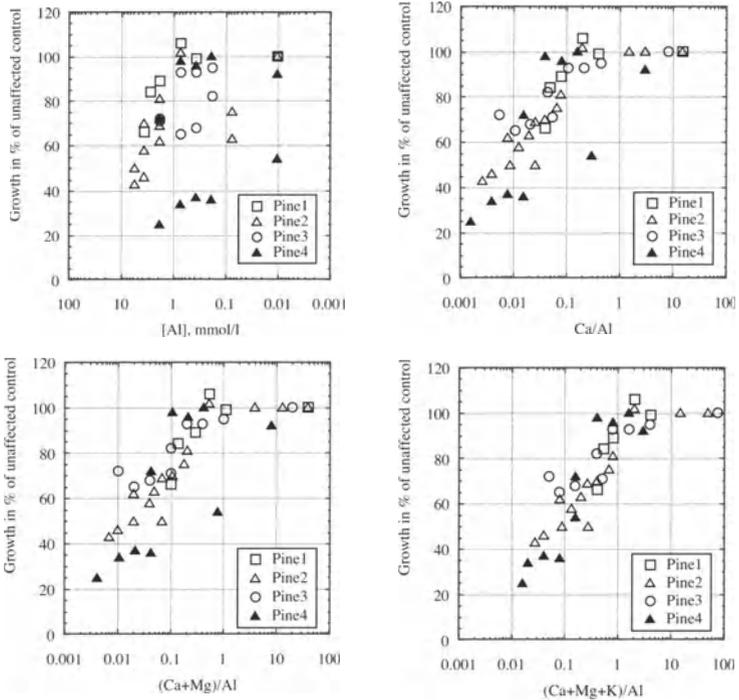


Figure 8.8: The four diagrams show the growth responses for Scots pine in relation to the Al concentrations in 4 experiments of Arovaara and Ilvesniemi (1990) and of Ilvesniemi (1992). The best consistency is obtained by relating response to the $(Ca+Mg+K)/Al$ ratio.

than two years was excluded in the figure. The remaining data show much greater consistency. The results of experiments by Abrahamsen (1984), indicate a slight additional antagonism of Ca in addition to the 1:1 of the BC/Al ratio. Fig. 8.9 shows the response of different types of pine to Al as observed in laboratory assays. Pines lie somewhere between the unspecific and the Vanselow response, with the exception of the data by Arovaara and Ilvesniemi (1990), which indicate a less sensitivity and a more unspecific response. The responses cluster, including plants with semi-quantitative data, into two groups with respect to Al sensitivity. The response data is for Scots pine (*Pinus sylvestris*) as reported by Arovaara and Ilvesniemi (1990), Göransson and Eldhuset (1991) and McCormick and Steiner (1978). By comparison, the response for such pine species as Armand pine (*Pinus armandii*), Masson pine (*Pinus massoniana*), Aleppo pine (*Pinus halepensis*), jack pine (*Pinus banksiana*), white pine (*Pinus strobus*), Scots pine (*Pinus sylvestris*), Longleaf pine (*Pinus palustris*), Monterey pine (*Pinus radiata*) and Loblolly pine (*Pinus taeda*) can be found in Sverdrup and Warfvinge (1993). Fig. 8.9 shows the bioassay response for European birch or Silver birch (*Betula pendula*). There is consistency between results for all birch species, despite differences in base cation concentrations and birch species (Sverdrup and Warfvinge 1993). Silver birch is more tolerant to Al than aspen. There is consistency between the results of Göransson and Eldhuset (1987) and the other results, despite fundamental differences in experimental design. Aspen (*populus tremuloides*) is shown in Fig. 8.9. Alder appears to be significantly more sensitive than Silver birch, European beech and oak. The data was taken from from McCormick and Steiner 1978, Steiner et al. 1980, 1984, and

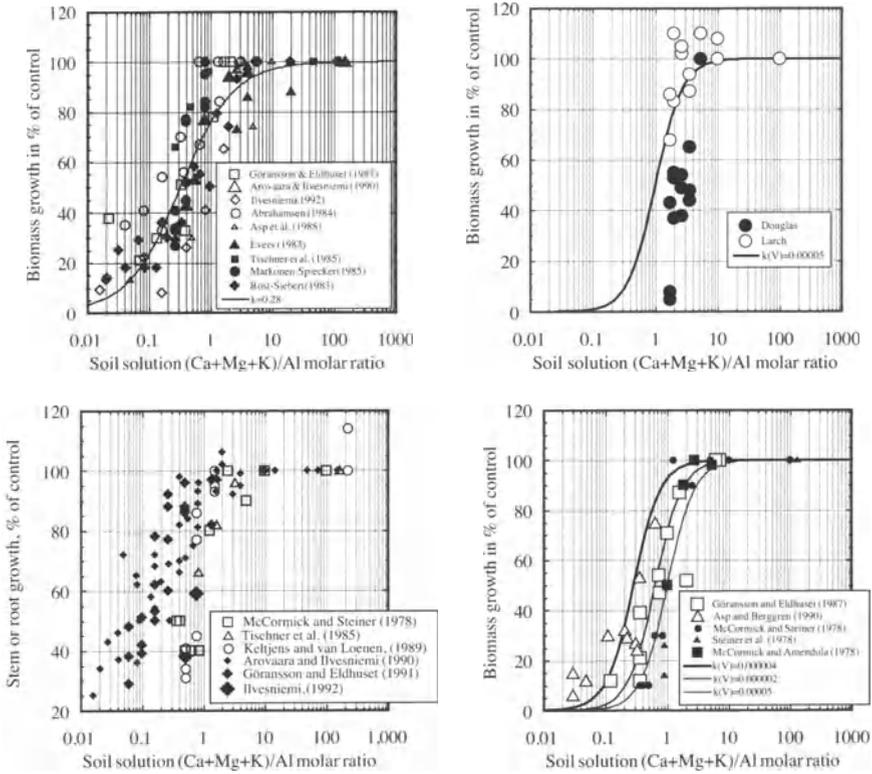


Figure 8.9: Growth response for Norway spruce (Norway spruce), Larch (*Larix decidua*), Scots pine (*Pinus silvestris*) and Silver birch (*Betula pendula*) in laboratory experiments.

McCormick and Amendola 1983. Fig. 8.10 shows a comparison between laboratory bioassay results for European oak (*Quercus robur*) and American red oak (*Quercus rubra*). Both tree species show the same response. Fig. 8.10 shows response data for European beech (*Fagus sylvatica*) and American beech (*Fagus grandifolia*) from laboratory bioassays in Germany and Sweden, as compared with the available field data. The comparison shows (two data points for beech, one point for oak and six points for orange) that limits based on laboratory data for beech can probably be transferred to field conditions without rescaling. Beech apparently follows the Vanselow response mechanism and shows little elasticity in its response to Al as compared with spruce. The data was taken from Rost-Siebert 1983, Asp and Berggren 1990 and the field data from Ulrich (1985). American beech appears to be slightly more tolerant to soil acidity than its European relative. Fig. 8.10 shows laboratory bioassay results for sugar maple (*Acer saccharum*) trees from the northeastern United States and from eastern Canada, a tree species used for the commercial collection of maple syrup. Both stem growth and root growth data are shown, indicating the same response. Sour orange (*Citrus aurantium*) and Japanese mandarin orange (*Citrus natsudaidai*) are shown in Fig. 8.10. The data originate from seedling experiments, younger trees and trees several decades old in a Japanese nursery (Fig. 8.11).

These data for orange are important since they include not only seedlings, but also fully grown trees age up to 67 years of age (Worku et al. 1983). They show that the data from laboratory experiments for orange can safely be applied to field conditions. Tab.8.9 lists English name, latin name, type of Al response, coefficient of the response function and BC/Al ratio at which growth has been reduced to 80% of normal for various spruce and pine species. Mycorrhiza is considered to be an important part of the root system of a tree, the tree and the mycorrhiza fungus living in a symbiosis. Mycorrhiza is generally regarded as a kind of integrated extension of the root system, and it may be as large or larger than the root system itself. The mycorrhiza system is

generally considered to be very important for the nutrient collecting capability of the tree, and anything that could affect the functioning of the mycorrhiza system would be of interest. Accordingly, a few experiments have been designed to study the effect of soil acidity of roots without mycorrhiza and trees with roots infected with mycorrhiza. The data available for Balsam fir (Entry et al. 1987) and Scots pine (Göransson and Eldhuset 1987) may seem to suggest that mycorrhiza is of no relevance for the response to Al whether the root is infected by mycorrhiza or not. Possibly, the opposite is the case, a tree infected with acid sensitive mycorrhiza may be more vulnerable than a tree without or with acid resistant mycorrhiza species. Entry et al. (1987) showed the effect of soil acidity and Al directly on mycorrhiza and other soil microorganisms. Similar results were also obtained for bacteria (Ohno et al. 1988). The response isotherm for mycorrhiza alone appears to be similar to that of Spruce and Pine (Figs. 8.12). Mycorrhiza may have a key role, they form important support systems for nutrient supply to the trees. Quantitative data to prove this point remain elusive, and very much about the nature of the role actually played by mycorrhiza remain hypothetical. It goes without saying that mycorrhiza is no charity organization, and whatever benefit the tree gets from the fungus, the fungus does this only as long as it benefits from it, thus at some cost to the tree.

Possibly, the tree can exploit the mycorrhiza for nitrogen and water, the cost would be that the tree will have to compete for phosphorus and and that the fungus would prey on carbon exudates from the root. If mycorrhiza would be more tolerant to Al than the tree, then it would be potentially able to offset the effect of Al. Unfortunately, this appear not to be so. Mycorrhiza appear to be as sensitive to Al as its host tree. Entry et al. (1987) investigated the response of the mycorrhiza associated with Balsam fir. The data available for Balsam fir and Scots pine does indicate that it is of no relevance for the response to Al whether the root is infected by mycorrhiza or not. Data from Göransson and Eldhuset (1991) show that there is no difference in response between trees infected with mycorrhiza and trees not infected with mycorrhiza (Fig. 8.12). The response isotherm for mycorrhiza alone appears to be similar to that of spruce and pine. We can therefore not expect mycorrhiza to protect tree from the effect of soil acidity. On the contrary, because mycorrhiza appears to be sensitive to aluminium, trees dependent on mycorrhiza may be more jeopardized by the effects of soil acidification.

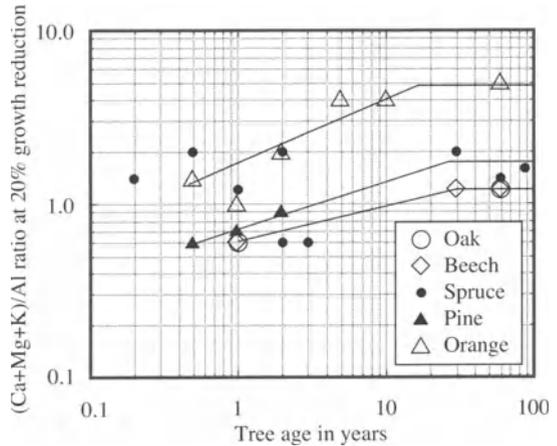


Figure 8.11: As can be seen from the data, the sensitivity can change over time as the plant become older. Orange, beech, oak and pine all appear to follow the same pattern, whereas spruce appears to show a different behaviour. All trees become more sensitive with age.

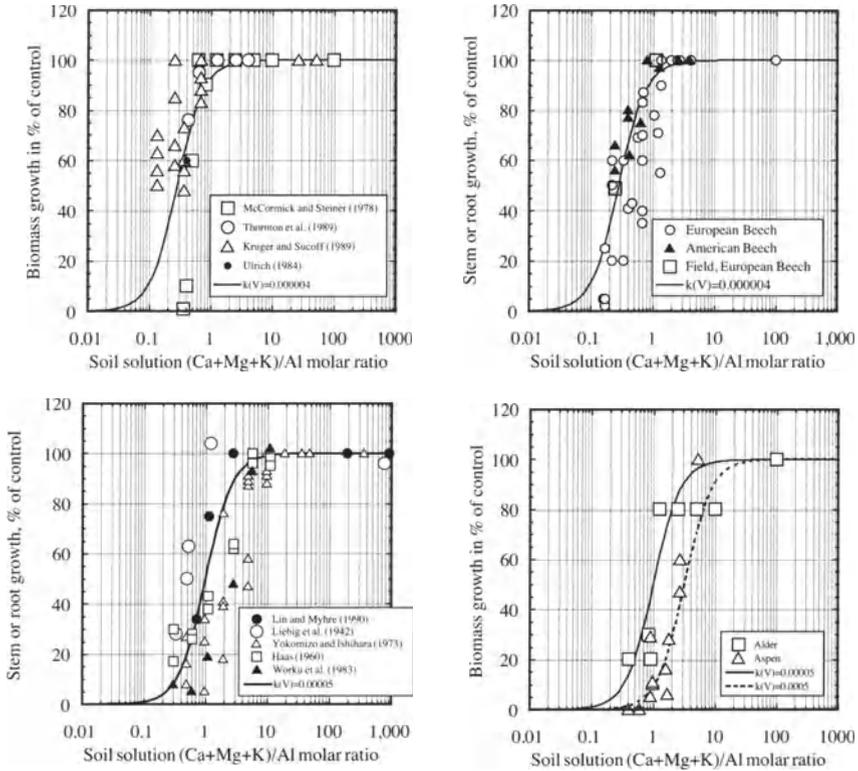


Figure 8.10: Top right: Laboratory experiments and field observations of growth decline of European beech (*Fagus sylvatica*) and American beech (*Fagus grandifolia*). Two data points with responses from field observations of growth decline of European beech (*Fagus sylvatica*) have been included and are shown by the open square symbol in the diagram. Top left: Growth decline of oak. German field data for oak has been included in the diagram (Black dot, Ulrich 1984). The field data show that the BC/Al concept holds under field conditions and that the result of laboratory experiments can be transferred to field conditions. Bottom left: Aspen (*Populus tremuloides*). Bottom right: Sweet orange (*Citrus sinensis*) and Japanese mandarin orange (*Citrus natsudaidai*) in lab and long term field experiments.

| English | Latin name | Reaction | K-value | BC/Al |
|-------------------|------------------------------|----------|-------------|-------|
| Sitka Spruce | <i>Picea sitchensis</i> | n=m=1 | K=0.1 | 0.4 |
| White Spruce | <i>Picea glauca</i> | n=m=1 | K=0.2 | 0.5 |
| Black Spruce | <i>Picea mariana</i> | n=m=1 | K=0.25 | 0.8 |
| Norway Spruce | <i>Picea abies</i> | n=m=1 | K=0.35 | 1.2 |
| Red Spruce | <i>Picea rubens</i> | n=m=1 | K=0.35 | 1.2 |
| Balsam Fir | <i>Abies balsamea</i> | n=m=1 | K=0.25 | 1.5 |
| Fraser fir | <i>Abies fraseri</i> | n=m=1 | K=0.35 | 1.2 |
| Western red cedar | <i>Thuja plicata</i> | Vanselow | K=0.0000001 | 0.09 |
| White Pine | <i>Pinus strobus</i> | n=3, m=2 | K=0.000002 | 0.5 |
| Western hemlock | <i>Tsuga heterophylla</i> | Vanselow | K=0.0000003 | 0.2 |
| Douglas fir | <i>Pseudotsuga menziesii</i> | Vanselow | K=0.0000004 | 0.3 |
| White pine | <i>Pinus strobus</i> | Vanselow | K=0.000002 | 0.5 |
| Aleppo pine | <i>Pinus halepensis</i> | Vanselow | K=0.000002 | 0.5 |
| Slash pine | <i>Pinus elliotii</i> | Vanselow | K=0.000002 | 0.5 |
| Armand pine* | <i>Pinus armandii</i> | Vanselow | K=0.000015 | 0.5* |
| Sand pine | <i>Pinus clausa</i> | Vanselow | K=0.000004 | 0.6 |
| Monterey pine | <i>Pinus radiata</i> | Vanselow | K=0.00008 | 0.8 |
| Scots Pine | <i>Pinus sylvestris</i> | n=3, m=2 | K=0.000002 | 1.2 |
| Scrub pine | <i>Pinus virginiana</i> | Vanselow | K=0.00002 | 1.2 |
| Pitch pine | <i>Pinus rigida</i> | Vanselow | K=0.00002 | 1.2 |
| Jack pine | <i>Pinus banksiana</i> | Vanselow | K=0.00003 | 1.5 |
| Loblolly Pine | <i>Pinus taeda</i> | n=3, m=2 | K=0.00002 | 1.5 |
| Larch | <i>Larix decidua</i> | Vanselow | K=0.00005 | 2 |
| Longleaf pine | <i>Pinus palustris</i> | Vanselow | K=0.00005 | 2 |
| Masson Pine | <i>Pinus massonii</i> | n=3, m=2 | K=0.0001 | 4 |

Table 8.9: Response type and estimated aluminium response coefficients for spruce, pine and other conifers. The BC/Al-limit was determined on root growth or whole plant growth, and set for laboratory results reduced to 80% of normal. *: field estimate.

8.4.4 Discussion

When located in the field the plants are subjected to the combined effects of different BC/Al values in the rooting zone. In the O- and E-layer, the BC/Al value may under moderate acidification be significantly different from the value in the B-horizon. Under laboratory conditions, the plants are generally exposed to uniform soil conditions, there being one BC/Al value for all roots. The laboratory experiments represent the effect of the BC/Al which the plant roots in a particular soil layer would experience, this need to be weighed together with results for all the layers in order to estimate what the plant as a whole would experience. One conclusion is that multi-layer models are preferable for sustainability assessments. The often stated conclusion that the Al concentrations required to induce tree response in experiments are seldom observed in the field is probably a misinterpretation of the experiments. When the differences between field and experiment base cation concentrations are taken into account, damaging Al concentrations can indeed be seen to occur in acid soils. It is a consistent pattern that the growth effect expressed as a function of the $(Ca+Mg+K)/Al$ ratio rather than of the Al concentration alone generally removes most of the differences between such studies on a given plant species. Some of

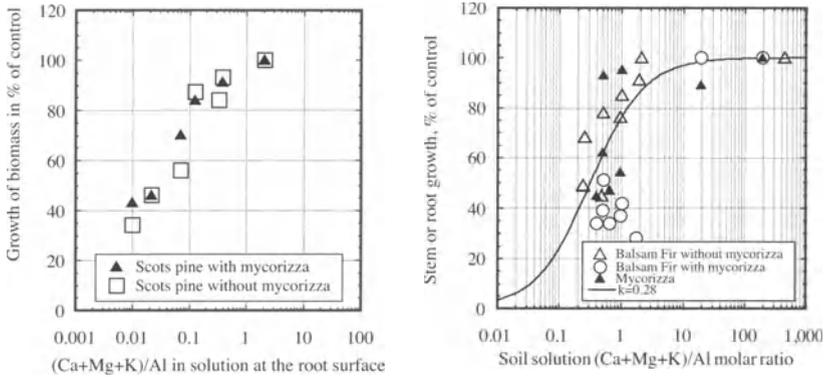


Figure 8.12: A comparison between experiments with Scots pine without mycorrhiza and Scots pine with mycorrhiza, showed no difference in the response of growth to soil solution Al (Göransson and Eldhuset 1987). There is a near 1:1 correspondence between the response to Al for Scots pine with and without mycorrhiza. It does appear that mycorrhiza cannot offer the plant extra protection against soil acidity. From the perspective of mycorrhiza being an important part of the root apparatus, this is a serious treat.

the studies revealed that the P/Al ratio may also be an important factor for growth, especially in soils with no or little excess production of P over growth demand (Asp and Berggren, 1990). Since P is an essential element and is growth rate regulating, similar effects on trees can be expected as well. The nitrogen sources for the plant may be important due to several different mechanisms, such as nitrification, ANC production through nitrate uptake by plants in contrast to the acidity produced by ammonium uptake or by plant variability. In several studies reporting the effect on Al-tolerance of the type of N source, the effect of acidity production from N transformation seem to be the most important factor in causing an effect. the conclusions are

- Soil acidity affects tree vitality mainly through changes in nutrient uptake capacity. Generally, the uptake of Ca, Mg and K seems to be most disturbed.
- There seems to be a difference between species regarding which nutrients are most strongly affected by Al^{3+} and H^+ , respectively. If the two stress factors act concomitantly, beech and oak are most sensitive to a combination of aluminium and H^+ stress.

8.4.5 Nitrogen and nutrient imbalance

Ulrika Rosengren, Ingrid Stjernquist and Gunnar Thelin

Up to the 20th century, Swedish forests developed under conditions of nitrogen limitation. The extra input of nitrogen from air pollution deposition was insignificant, as can be illustrated by an investigation from the beginning of the 1950s showing that the maximum contribution of nitrate-N through deposition was 0.8-0.6kg/ha,yr (Emanuelsson et al. 1954). This can be compared with the current values of 10-20 kg N ha⁻¹yr⁻¹ in southern Sweden. The increased nitrogen availability affects trees in two principal ways:

1. Through nitrogen as a growth factor
 - (a) where it increases the water demand of the tree through an increased growth and a change in the root to above-ground biomass ratio, causing an increased risk of

| Plant species | Latin name | Reaction | K-value | BC/Al |
|-----------------|------------------------------|------------|------------|-------|
| Teak | <i>Tectona grandis</i> | Vanselow | K=0.000004 | 0.6* |
| Sugar maple | <i>Acer saccharum</i> | Vanselow | K=0.000004 | 0.6 |
| Oak | <i>Quercus robur</i> | Vanselow | K=0.000004 | 0.6 |
| Red oak | <i>Quercus rubra</i> | Vanselow | K=0.000004 | 0.6 |
| Pin oak | <i>Quercus palustris</i> | Vanselow | K=0.000004 | 0.6 |
| Silver Birch | <i>Betula pendula</i> | n=3, m=2 | K=0.000006 | 0.7 |
| American beech | <i>Fagus grandifolia</i> | Vanselow | K=0.000004 | 0.6 |
| Beech | <i>Fagus sylvatica</i> | n=3, m=2 | K=0.000004 | 0.6 |
| Peach | <i>Prunus persica</i> | Unspecific | K=0.4 | 1.4 |
| Black Alder | <i>Alnus glutinosa</i> | Vanselow | K=0.00005 | 2 |
| Paper birch | <i>Betula papyrifera</i> | Vanselow | K=0.00005 | 2 |
| Gray birch | <i>Betula populifolia</i> | Vanselow | K=0.00005 | 2 |
| Yellow birch | <i>Betula alleghaniensis</i> | Vanselow | K=0.00005 | 2 |
| Sour orange | <i>Citrus aurantium</i> | Vanselow | K=0.00005 | 2 |
| Eucalyptus | <i>Eucalyptus gummiifera</i> | Vanselow | K=0.00006 | 2.8 |
| Mandarin orange | <i>Citrus natsudaikai</i> | Vanselow | K=0.0003 | 3 |
| Rhododendron | <i>Rhododendron ponticum</i> | Vanselow | K=0.0004 | 4.5 |
| Crack willow | <i>Salix fragilis</i> | Gapon | K=0.08 | 5 |
| European alder | <i>Alnus glutinosa</i> | Vanselow | K=0.0002 | 5 |
| Aspen | <i>Populus tremula</i> | Vanselow | K=0.0005 | 6 |

Table 8.10: *Response type and estimated aluminium response coefficients for different species of deciduous trees and bushes ordered according to relative sensitivity. The BC/Al-limit represents a growth reduction to 80% of normal. * BC/Al=0.35 based on a field value for stem growth. n.d. represents values derived from a combination of the quantitative data in this study and the semi-quantitative data of Cronan et al. (1989), Kowalkowski (1987) and Ulrich (1985) as well as unpublished data.*

drought stress.

- (b) where it decreases frost-hardiness during the winter period causing an increased risk of frost injuries at low winter temperatures or as a result of sudden frost periods during the spring.
- (c) where it increases the risk of nutrient imbalance in the trees
- (d) where it contributes to a reduction in the soil-nutrient pool through an increase in uptake.

2. Through effects on the soil environment

- (a) where it increases the soil acidification and the leaching of Ca, K and Mg from the root zone.
- (b) where it hampers the external mycorrhiza development and thus decreases the nutrient uptake capacity of the tree.
- (c) where it increases the risk of nitrate leaching from the soil to water courses.

Reaction of the ecosystem to excess nitrogen

Since nitrogen availability is traditionally the growth-limiting factor in terrestrial ecosystems and in Swedish forest trees, an increased nitrogen input to the ecosystem can be expected to increase production (Tamm 1991). On the other hand, if the forest has received nitrogen beyond the point of nitrogen saturation, nitrogen loss from the ecosystem is likely to occur as may visible damage to the trees (Fig. 8.13). During the last two decades, a N-saturated ecosystem has been defined in different ways. The three definitions most frequently used are

- An ecosystem in which primary production will not be increased further by an increase in the supply of N (Nilsson 1986).
- An ecosystem in which N-losses approximate or exceed the inputs of N (Ågren and Bosatta 1988).
- An ecosystem in which the availability of inorganic N is in excess of the combined total plant and microbial nutritional demand (Aber et al. 1989)

The different criteria used has confused the discussion about at what nitrogen level the risk of forest damage increases. The first two definitions can allow considerable amounts of nitrogen to leach out of the forest soil before the stand is regarded as having reached the point of N-saturation. The third definition reflects a sustainability perspective, the idea that the nitrogen availability should not exceed the demand of the organisms living in the forest and that no nitrogen should leach from the forest ecosystem and have a negative impact on other ecosystems. From a forest health perspective, nitrogen availability is clearly in

excess if it leads to nutrient imbalance in the trees. Even if tree growth increases, risk of stress from drought, frost, pathogens and pests may develop and materialize when situations are unfavourable. The C/N ratio in the topsoil has been suggested to be a useful indicator of the risk of nitrogen-leaching in spruce forests and thus, of the degree of N-saturation. The threshold value appears to be a C/N ratio of approximately 25 (Gundersen et al. 1998). Below

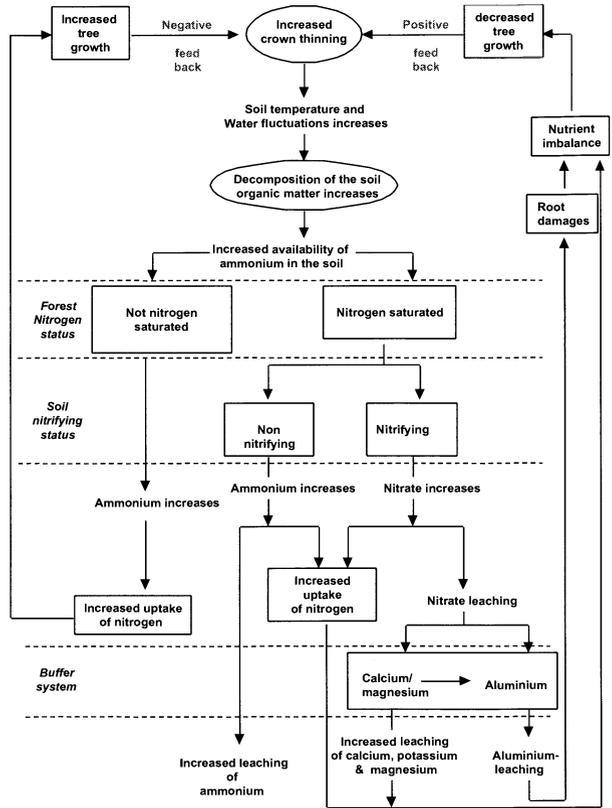


Figure 8.13: Relationship between nitrogen saturation status and forest damages, according to Berg et al. 1997. The nitrogen deposition adds directly into the system by increasing nitrogen concentrations. For more detailed explanations see text.

this value nitrate leaching is probable and the amount of nitrate in the outflux water increases exponentially as the ratio decreases.

Excess nitrogen and stand age

Young stands appear to be able to receive large amounts of N before any restrictions on growth, such as limitation by another nutrient, appear. In Norway spruce stands in southern Sweden, the annual growth rate increases to approx. 40 years, depending on the site conditions, and then declines (Thelin 2000). It appears that the fast-growing young stands are able to efficiently absorb all deposited N. Both the growth rate and the nutrient requirements change over time as a stand develops (Miller 1995, Kimmins 1997). Stands more than 40 years old, however, appear to behave quite differently from younger stands, showing clear indications of being limited by K or P rather than by N (Thelin et al. 2001b). According to Miller (1995), nutrient deficiencies are most likely to appear in young stands when crowns are being constructed, since fast-growing young stands depend on soil reserves of nutrients to a greater extent than older stands do, in which a large share of the nutrient demand is satisfied by retranslocation (Miller 1986). Miller (1995) adds that "if deficiencies are rectified prior to canopy closure there are then unlikely to be any further nutritional problems providing that nothing occurs to disrupt one or more of the nutrient cycles". This is probably what has happened in the Norway spruce stands in southern Sweden that for decades have received an enhanced N and S deposition. Apparently, when stands reach the age of approximately 40 years soil pools of nutrients can no longer sustain the high N-induced growth rates. It should be emphasized that, although N additions may increase forest growth in large parts of southern Sweden in the short run, this is clearly not sustainable in the long run.

Nutrient imbalance

The loss of nutrients through leaching and an increased removal of harvested biomass both of them due to a higher growth induced by N, may lead to an imbalance between nutrients in soil and trees. The term "nutrient imbalance" can be used to describe

- An imbalanced nutrient situation within a tree. This is manifested by an absolute or relative deficiency of one or more nutrients.
- Nutrient depletion of the soil, i. e. when the concentrations of nutrients decrease below what would be sustainable for long-term forest production.
- A deficit of mineral nutrients within the forest ecosystem due to an imbalance between the inputs and outputs of nutrients. This can be described as the weathering plus deposition inputs being smaller than leaching plus uptake caused by stem growth (Sverdrup and Rosén 1998).

The third description may be regarded as the precursor to 1 and 2. Deficits may exist for a long period of time before soil or tree nutrient imbalance develop. Considering the well documented deterioration of forest health and the increase in soil acidity, the increase in forest productivity could be seen as a paradox. However, when this paradox is examined more closely, it becomes evident that one can reasonably expect that in a system in which N is growth-limiting both growth and acidification should increase when N availability increases. High growth rates can be maintained as long as no other limitation on growth replaces the nitrogen limitation. This will not appear before the pools of available mineral nutrients become too small to satisfy the nutrient demands of the tree or before the nutrient uptake is otherwise impaired. For example, an excess of nitrogen, since it inhibits the development of external mycorrhiza, leads to a decrease in the nutrient uptake capacity of the tree. (Wallander 1995, Kårén and Nylund 1997). This nitrogen effect may be either direct or indirect. Further studies are needed to better understand the exact mechanisms involved. In addition, nutrient imbalance may affect tree vitality through

- An increased drought sensitivity due to a decrease in the foliar ratio of K, Ca and P to N (Rosengren-Brinck and Nihlgård 1995)
- A decreased frost sensitivity due to a decrease in the availability of K, Ca and Mg.
- An increase in the risk of attack by aggressive pathogens and pests, e.g. *Armillaria* (Shaw and Kile 1991). The importance of a specific element depends on tree species.
- An increase in the risk of growth malformations due to a decrease in the foliar ratio of B and Cu to N

The nutrient threshold values for leaves and needles used as indicators for forest vitality are given in chapter 8.7. The deficiency levels for foliar nutrient concentrations, as well as the target ratios to N, are well defined for spruce and pine, whereas the threshold values for important deciduous tree species such as birch, beech and oak are more uncertain at present.

Foliar nutrient concentration and stress

To evaluate the nutrient demand of the tree and diagnose possible deficiencies, the nutrient concentrations and corresponding plant growth need to be known. A deficiency has been defined as the concentration of nutrient X when growth has decreased by >10%. These types of studies are generally carried out in nurseries (e.g. Binns et al 1980) or in hydroponic systems by use of sprouting seedlings (e.g. Ingestad 1988). However, since it has been argued that the nutrient demand of a tree may differ depending upon whether the tree is a seedling, a young plant or a mature tree, another way of defining both deficiencies and optimum nutrient levels has been to screen the nutrient status of mature trees in the field displaying no visible damages, (e.g. Bergmann 1988). In efforts to understand differences in nutrient elasticity between species, data found in literature on nutrient concentrations in the foliage of mature trees was examined and frequency distribution diagrams being created. This was done for four important Swedish forest tree species Norway spruce (*Picea abies*), birch (*Betula pubescens*), beech (*Fagus sylvatica*) and oak (mainly *Quercus robur* but also *Quercus petraea*). Only data from stands with no visible damage were employed; neither solitary individuals nor trees from treated experimental plots were included.

The data used in the analysis was taken both from literature and in some cases from unpublished data from the SUFOR research projects

- Birch: mainly Swedish information, *Betula pubescens* being included,
- Norway spruce: mainly information from southern Sweden
- Beech: European information, Swedish information from Scania
- Oak: European information, Swedish information from southern Sweden. Both *Quercus robur* and *Quercus petraea* was used.

These frequency diagrams, besides showing between-species differences in nutrient elasticity, allow threshold values to be estimated, i.e. the point where the curve starts leveling off. It can be argued that this point can be interpreted as representing one way of setting critical levels for nutrients. As can be seen in Fig. 8.15, the frequency graph of N, Ca, K, Mg and P concentrations in the foliage shows two different patterns dependent of nutrient. Over the whole range of soil fertility, the concentrations of K, Ca and N in spruce needles are well below those for the deciduous species, whereas for Mg and P, birch differs from the other species by its wider range in concentrations, i.e. its greater nutrient elasticity. The Mg graph shows both higher as well as a wider range of foliar nutrient concentration for birch. For P, the lower end of the curve is similar for all the four tree species whereas the upper part of the graph clearly singles out birch, again with both higher as well as a wider range of foliar nutrient concentration. The

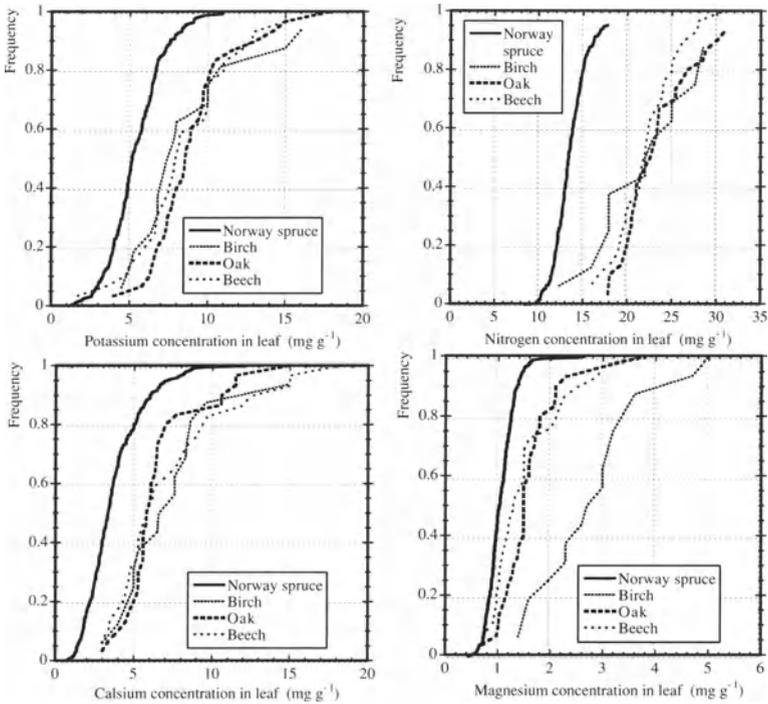


Figure 8.14: *Top:* Frequency graph of the foliar concentrations of N and K for oak, beech, birch and Norway spruce in Europe. *Bottom:* Frequency graph of foliar concentrations of Ca and Mg for the same species.

threshold value of 1.8 for P indicated by the frequency graph (where the curve makes a "knee") can probably be an artefact. Thus, more studies are needed to determine the true frequency distribution for birch. One explanation for the generally higher P and Mg concentrations in birch leaves could be that a fast-growing species as birch require more Mg for photosynthesis as well as more P for energy transport than the more slow-growing species. The frequency graphs of the K, Mg and P ratios to N also show patterns that differ, depending on element and species, Fig. 8.14. The K/N frequency graph is for oak, beech, and Norway spruce basically the same, probably indicating that, these species are similar in their use of potassium in physiological processes. Hence, one and the same K/N ratio appears to be the most suitable indicator to evaluate stress in forest stands, independent of stand characteristics. On the other hand, the graph for Mg/N follows the life strategies of the studied species. For birch, as a fast-growing species, the ratio is higher than for the slow-growing spruce, oak and beech. This tendency holds both for Mg and for the Mg/N ratio. The frequency graphs for P/N are more complicated to evaluate. The P/N graph for spruce shows higher ratios than for any of the other species generally, even when it is compared with that of birch. An explanation for this may be that spruce requires a higher proportion of P in order to store energy for producing costly defense substances such as resins and phenols. Future analysis of the situation for pine may be able to cast more light on the tenability of this hypothesis. The Mg/N and P/N ratios may potentially be used as stress indicator when comparing the vitality status between different tree species. It has been argued that using the ratio of nutrient X to nitrogen is a more reliable way of diagnosing nutrient imbalances and thus deficiencies (e.g. Ingestad 1979c), due to the fact that

nitrogen is the most growth promoting nutrient and it controls growth to large extent.

Within a wide range, the concentration per se is not considered to be essential for the "vitality" of a tree the proportions of elements relative to nitrogen being at least as important (Linder 1995). Linder (1995) also suggested that the relative proportions of nutrient X and nitrogen are similar for the all higher plants. However, Interpretation of the frequency graphs contradict the statement just cited. These data show that for the ratio of potassium to N the same threshold value may well be used for each of these four tree species. Beech, oak and Norway spruce seems, on the other hand, to have the same threshold value for Mg/N, whereas the value for birch should be higher. With regard to P/N ratios, species specific values should be set for Norway spruce and birch, while the same threshold value may be used both for oak and beech. Further studies are needed to verify these findings.

Nitrogen as a predisposing factor for multiple stress

In chapter 8.1 the current predisposing stress factors in southern Sweden were discussed. Many of these factors, like nitrogen and water, are essential and limiting for tree growth in low concentrations. In his stress concept, Godbold (1998) stated that tree adaption to constraints is carbon expensive. This is also true of constraints which are factors essential for growth if the concentration is above the optimal value. Such long-lasting constraints cause carbon costs for the repair of damage and for reducing physiological stresses. The carbon costs are species-dependent but are probably also dependent on age, due to the size of the nutrient pool, which can recirculate within the plant. Such a period of "self cannibalism" is theoretical longer for mature trees than for seedlings and is longer for seedlings with large seeds than for those with small seeds. Fig. 8.16 shows on top the effect of nitrogen fertilization ($150 \text{ kg N ha}^{-1}\text{yr}^{-1}$) and soil acidification on the K and P content of beech seeds (Stjernquist 2002b). The middle figure shows P uptake in 2 yr old beech seedlings grown under stress conditions. In the figure $\text{NO}_3\text{-N}=100 \text{ kg Ca}(\text{NO}_3)_2\text{ha}^{-1}\text{yr}^{-1}$; $\text{NH}_4\text{-N}=100 \text{ kg NH}_4\text{NO}_3\text{ha}^{-1}\text{yr}^{-1}$; $\text{acid}=\text{pH}_{\text{KCl}} 3.30$; $\text{mod. acid}=\text{pH}_{\text{KCl}} 3.87$; D = drought treatment, $n=10$. (Stjernquist 2002a). The bottom diagram shows number of shoots remaining and missing related to total number of shoots on branches taken from the 7th whorl from Norway spruce (Rosengren-Brinck and Nihlgård 1995). A changed carbon allocation pattern may also influence the possibility of the tree to withstand other stress factors. When nitrogen availability increases from the limitation to the optimal value for tree growth, the shoot to root ratio increases. Thus, constraints that negatively affect the nutrient uptake capacity, such as ozone and soil acidity, should have synergistic effects when combined with a high nitrogen availability. This has been studied in field experiments with use of the most probable stress factors in southern Sweden, excess of nitrogen, soil acidification and drought. Below, three examples are discussed which concern three different

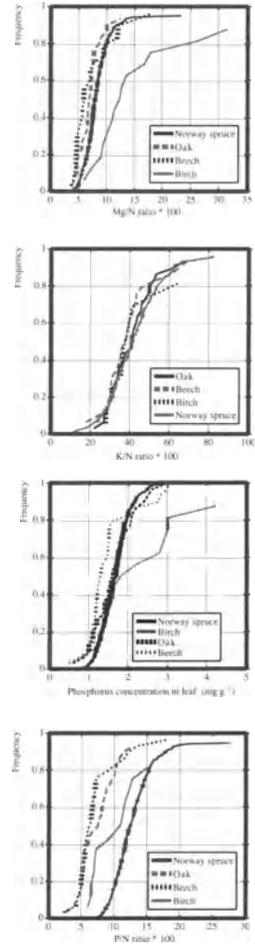


Figure 8.15: Frequency graph of the foliar nutrient ratios to N (Mg/N, K/N and P/N) for oak, beech, birch and Norway spruce in Europe and the frequency graph of the foliar concentration of P for the same species.

life stages, mature tree, seedling and seed.

Seed

In Sweden, deciduous trees are mainly regenerated through natural seed fall. Species with large seeds, like beech and oak, have a large nutrient pool at germination, allowing the seedlings to compete with the forest floor vegetation through growth and root development. For beech, for example, the relative growth rate of seedlings during the second year is higher than the relative growth rate of the leaves. This is probably due to a residual effect of the large nutrient and carbon reserves in the seed, which is pooled to the development of the stem and main roots during the first year. On the very acid clayey and silty soils in southern Sweden, impoverished of important cations through leaching, the beech and oak seedlings are strongly dependent on the nutrient content of the seed during the first vegetation period. Thus, the seed condition, defined as the nutrient content of the endosperm, is an important factor for the future survival of the plant. In northern Europe, the filling of beech seeds starts in August and continue until leaf fall (Röhrig et al 1978). Since in southern Sweden, beech leaves normally are functional until the beginning of October (Staaf and Stjernquist 1986), the time for seed filling is theoretically two months. In this area, the air pollution deposition, however, causes early senescence of the leaves and a changed crown architecture. The effects, single and in combination, of excess of nitrogen and soil acidity on beech endosperm mass and nutrient content were studied in a field experiment in mature beech stands. The excess of nitrogen was performed through fertilizing with 150 kg N/ha,yr for five years. The results show that excess of nitrogen decrease the K content of the seed by 8%, both on acid and on moderately acid soil. On moderately acid soil the P content also decreased by 10% (Fig. 8.16). Beeches growing on acid soils was found to produce a significantly lower endosperm mass than those growing on moderately acid soils, the difference being as high as 17% (Stjernquist 2002b). These results indicate that regeneration of beech in southern Sweden is likely to be negatively affected by the current air pollution conditions. The combined effects of excess nitrogen and acid soil conditions, together with low and imbalanced nutrient content, results in reduced seed size with a smaller relative content of P and K (Stjernquist 2002b).

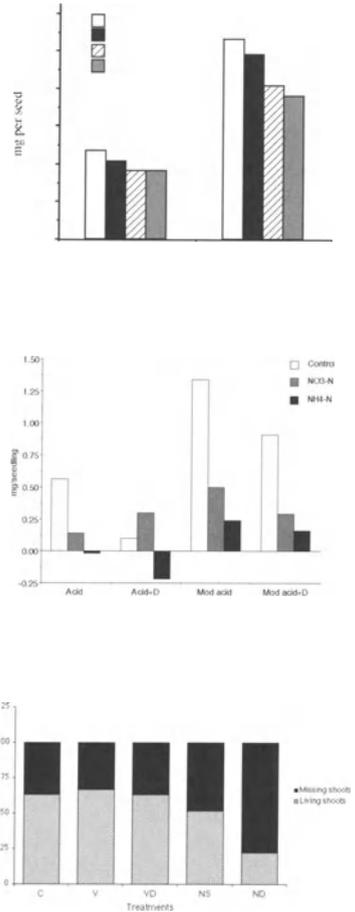


Figure 8.16: Top; Effects of nitrogen fertilization and soil acidification on the K and P content of beech seed. Middle; P uptake in 2 yr old beech seedlings grown under stress. Bottom; Number of shoots remaining related to total number of shoots on branches taken from the 7th whorl from Norway spruce.

Seedlings

The reaction of small plants was studied in an experiment with beech seedlings established under a beech shelter-wood. The seedlings were grown for two years in an acid till (pH_{KCl} 3.3) and a moderately acid till (pH_{KCl} 3.9). The nitrogen treatments were $100 \text{ kg N ha}^{-1}\text{yr}^{-1}$ of $\text{Ca}(\text{NO}_3)_2$ and NH_4NO_3 respectively. Drought stress was applied during the second vegetation period from May to September, the individuals growing in moderately acid soil and not subjected to any stress factor were used as control. The results showed N-saturated conditions to hamper the growth and development of beech seedlings independent of soil acidity. Nitrogen available as $\text{NH}_4\text{-N}$ had the most negative effect. Nitrogen excess also decrease the total uptake of P, Mg and K in the plants during the first two years (Stjernquist 2002a). When nitrogen excess was combined with low pH and with drought respectively, the N/low pH combination was found to have the strongest influence on growth and nutrient uptake. Under such circumstances, the relative growth rate of the leaf during the second year was zero or even negative in the case of the $\text{NH}_4\text{-N}$ treatment. The joint stress of N and low pH decreased P, Mg and K uptake still further. For beech seedlings, the P balance seems to be crucial. High availability of $\text{NH}_4\text{-N}$ in an acid soil causes a negative total P uptake in the 2 year old plants compared with the P pool in the seed.

Mature trees

A study of the effects of multiple stress on mature trees was conducted within the "Skogaby project" in which 30 year old Norway spruce (*Picea abies*) were grown under different nitrogen and water supply. The treatments used were control (C), ammonium sulphate (NS) where $100 \text{ kg N ha}^{-1}\text{yr}^{-1}$, N-free fertiliser (V) whereas all macro nutrients except for N were applied in a single dose. These three treatments were later combined with drought to create three additional treatments: drought (D), ammonium sulphate + drought (ND) and N-free fertiliser + drought (VD). The relation between nitrogen availability and tolerance for drought was studied by examining the flushing of new shoots and needle loss. Whereas NS treatment resulted in an increase in N but a decrease in the P/N, K/N and Mg/N ratios in the needles, use of N-free fertiliser resulted in an increase in the P/N and Ca/N ratios (Rosengren-Brinck and Nihlgård 1995). When drought was added (D, ND and VD) the nutrient ratios remained relatively unchanged as compared with C, NS and V. Thus, mature trees appeared to be very efficient in maintaining a stable internal nutrient balance. However, as a response to the nitrogen+drought treatment, the trees did stop the expansion of new shoots to a greater extent than trees from treatment VD (Fig. 8.16). In conclusion, nitrogen status had a substantial effect on drought resistance which could be seen in the differences in the flushing of new shoots rather than in changes in the nutrient ratios in the needles (Rosengren-Brinck and Nihlgård 1995).

Conclusions

It is evident that an imbalanced nutrient status reduces tree vitality and predisposes the trees to other kinds of stress. From a broad European perspective, it has been shown that inputs of excess nitrogen to forest ecosystems plays a major role in forcing forest ecosystems towards nutrient imbalance. It has been argued that the existing nitrogen deposition is no threat to Swedish forests (Binkley and Högberg 1997). We have provided several examples, however, where forests in southern Sweden suffer from an excess of nitrogen. This discrepancy in opinion may, in part, be owing to the fact that Swedish forests covers a large latitudinal span in which they are exposed to considerable differences in climate, as well as in nitrogen deposition, where the deposition decrease from southwest to northeast. Clearly, it is dangerous to generalise forest response to nitrogen deposition considering the gradients involved. Nutrient imbalance caused by nitrogen and acidification poses serious and increasing threats to the health and long-term productivity of southern Swedish forests. Although adaptive management and nutrient

compensation by base cations may improve the situation, they will remain no more than stop-gap measures. Accordingly, it is of utmost importance that N deposition to the forest ecosystems of southern Sweden is speedily reduced. This requires, from the Swedish perspective, a continued international effort to reduce emissions within the CLRTAP framework.

So far the effect of an excess of available nitrogen in southern Sweden is most evident as a predisposing stress factor which - in combination with frost, drought and soil acidification - causes stem damage, nutrient imbalance in the foliage and increased sensitivity to drought and storm. At sites with forced high production rates, e.g. spruce monocultures, the nutrient imbalance effect is most evident and unlikely to be sustainable in the long-term. However, differences in age and developmental stage, as well as in regeneration strategies and nutrient elasticities between tree species, are likely to determine the extent to which the identified effect will manifest itself in the field.

8.5 Stress by pathogens and pests

Felix Heintzenberg and Ingrid Stjernquist

8.5.1 Pest and pathogens in Swedish forests

Periodical outbreaks of insects and pathogens, which are well known phenomena and have been recorded by Swedish forest authorities for the last two centuries, often result in substantial economic loss for forest owners. Local and regional outbreaks of insects have been reported since the middle of the eighteenth century (Lekander 1950). These reports show that the most damaging pest and pathogen species have been the same during the period covered by the records. In Sweden, which is dominated by conifers, these species are *Hylobius abietina*, *Ips typographus*, *Pitogenes chalcographus*, *Tomicus piniperda*, *Tomicus minor*, *Hylastes cunicularius*, *Neodiprion sertifer*, *Bupalus piniarius*, *Lymantria monacha*, *Lymantria dispar*. The most common pathogens are root rot (*Heterobasidion Gremmeniella abietina*, *Lophodermium seditiosum*, *Melampsora pinitorqua* and *Armillaria spp.* (Skogsstatistisk Årsbok 2000)).

Knowledge of how to control insect and pathogen outbreaks by means of various forest management practices is also well established and includes clear cuts, removal of biomass, and the avoidance of thinnings or clear cuts during the spring, summer and autumn. High biomass concentrations after storm felling and harvest increases the risk in losses of productivity. Some of the most important species attack trees of decreased vitality, e.g. *Ips typographus*, *Pitogenes chalcographus*, *Tomicus piniperda*, *Tomicus minor* and *Armillaria spp.* For other species a suitable climate, temperature and humidity, during specific time periods is the triggering factor for population growth.

To prevent large outbreaks of insects and pathogens, the Swedish Forestry Acts have, since 1979, included regulations for tree harvest as well as for the removal and storage of timber and biomass. Forest owners and forest companies are responsible for ensuring that measures to inhibit or reduce insect breeding grounds and infestations be carried out. To control outbreaks, the National Forest Inventory has monitored the development of the most aggressive species.

In spite of a relatively effective insect and pathogen control, new types of forest damages have developed in southern Sweden during the last two decades, the most evident being oak decline and resin flow. The resin flow phenomena was first identified in southwest Sweden in the late 1980s and the oak decline in 1987 in the same area (Barklund et al. 1995, Sonesson 2001). Oak decline in Europe is connected with increased attack by *Phytophthora spp* and, for trees with resin flow, the presence of pathogen species has increased, especially *Tryblidiopsis pinastris* (Barklund et al. 1995). These types of forest damages are probably associated with environmental changes caused by anthropogenic stress factors, but the direct relationships are yet inadequately known. In Europe, forest areas with a high deposition of nitrogen have shown an increased infestation of foliage by grazing parasites (Flückiger and Braun 1999).

The new kind of forest damages in Sweden have highlighted the need for better knowledge of the relationship between nutrient imbalance in the soil, high deposition of nitrogen, high concentrations of ground-level ozone and a future climate change and the size of the insect and pathogen populations. This chapter will discuss the importance of nutrient imbalance as a triggering factor for increased attacks by pests and pathogens.

8.5.2 Effects of nutrient imbalances on pest and pathogens

Introduction

The effects of nutrient imbalances on the defense capacity of woody plants against pests and pathogens has long been overlooked. However, during the last two decades, more evidence has been presented for there being a connection between the increased amount of nitrogen in forest soils and the occurrence of attacks by fungi and herbivores. The current knowledge of the connection between nutrient imbalance, pathogens and pests will be reviewed in this chapter. An increased amount of nitrogen in forest soils results in an increase in tree growth, which requires large amounts of carbohydrates, produced during photosynthesis. Carbohydrates are used not only for tree growth and maintenance but also for the synthesis of carbon-based secondary metabolites, such as terpenes and phenolics. These are used as defense substances and are positively correlated with the C/N ratio of the plant (Bryant et al. 1983). Protective chemicals against pests and pathogens include phenols, tannins and lignin. Through its adhesive characteristic, resins are important in counteracting insect attacks. They also have antifungal properties by compounds that are directly toxic to animals and fungi. For example, monoterpenes and alkanes are toxic to the fungi transmitted by bark beetles (Kozłowski and Pallardy 1997, Farrel et al. 1991) The costs for synthesizing one gram of secondary metabolite of different defense substances were summarized by Gulmon and Mooney (1986), and estimated to be 2.6 g CO₂ for phenolic resins and 4.7 g CO₂ for terpenes. The production of defense substances is thus costly compared with the costs of leaf growth, which is given as 1.9-2.7 g CO₂ per gram of foliage. The costs of defense are considered as being a combination of the direct carbon costs and indirect costs, including future reductions in plant growth and reproduction (Gulmon and Mooney 1986).

Carbon nutrient balance - growth or defense

When the C/N ratio in the tree decreases, fewer carbohydrates are allocated to C-based secondary metabolites (Bryant et al. 1983). During extensive growth, secondary metabolism may therefore be limited (Gulmon and Mooney 1986, Chapin 1991). This causes a trade-off situation for the tree of either growing or defending itself against herbivore and pathogen attacks. Trade-offs between primary and secondary metabolism have been documented in cell cultures (Lindsey and Yeoman 1983). When nitrogen access is favourable, vegetative growth generally receives a resource priority over secondary metabolism and storage (Chapin et al. 1990, Waring and Pitman 1985). This has been defined as the Carbon Nutrient Balance (CNB)-hypothesis (Bryant et al. 1983) and is strongly supported by the results of a variety of studies. Herms and Mattsson (1992) stated that plant defense might decrease when the nitrogen content increases due to fertilization. Their statement was supported by Flückinger and Braun (1998), who showed that insect attacks (genus *Cinaria*, *Sacchimpanthes*) on *Picea abies* increased with an increase in N-fertilization.

Similar statements and investigations were made for a number of other trees. In beech, *Fagus sylvatica*, the total phenol concentrations in leaves significantly decreased by nitrogen fertilization (Balsberg-Påhlsson 1992). Flückinger and Braun (1998) showed that parasite attacks (genus *Phyllaphis*) increased with increasing N fertilization. Changes in the concentration of defense substances in *Picea sitchensis* as a result of changes in the root/shoot ratio due to nitrogen fertilization have been reported, the concentrations of the defense substances being highest at low nutrient and high light treatment (Wainhouse et al. 1998). Larsson et al. (1986) studied

changes in the amounts of phenolic compounds in willow *Salix dasyclados* leaves as a result of N fertilization and shading. At high N and low light, the leaves contained low concentrations of phenolic compounds owing to fewer carbohydrates being available for the synthesis of defense substances. Leaves, which contained low amounts of phenolics were preferably consumed by beetles *Galerucella lineola*. In the same experiment, high N and high light treatment resulted in high amounts of phenolic compounds and a low consumption of the leaves by beetles. These results confirm the hypotheses that the production of defense substances depends on the availability of carbohydrates, which in turn depends on growth and photosynthesis. In *Salix alaxensis*, the phenolic concentration was positively correlated with the concentration of soluble carbohydrates in the twig. In the experiments, fertilization and shading were found to reduce the amounts of carbohydrates and phenolics (Bryant 1987a). In *Salix eriocephala*, fertilized trees were found to be more susceptible to attacks by insect herbivores, e.g. a leaf chewing beetle *Popilia japonica*, than non-fertilized ones (Orlans and Floyd 1997).

Bryant et al. (1987b) conducted experiments which demonstrated that birch *Betula resinifera* was more exposed to grazing by herbivores when growth was increased by fertilizing with nitrogen and phosphorus. Shading decreased birch growth and increased the twig palatability. These results support the hypothesis that an increase in nitrogen availability supports carbon-demanding tree growth and at the same time disfavours the carbon-demanding synthesis of defense substances such as phenols. The hypothesis is further confirmed by the observation that plants growing on nitrogen-deficient soils appear to produce increased quantities of allochemicals (Bryant et al. 1983).

Nutrient imbalances and pathogens

Magnesium, potassium, calcium and phosphorous are necessary nutrients for plants (e.g. Nihlgård 1985). An imbalanced ratio of cations to N is considered to be an important factor for the successful infection of the host by a pathogen. High deposition of ammonium to forest soils hampers cation uptake (France and Reid 1983), resulting in leaching of base cations from the upper root horizon (e.g. Nihlgård 1985). This is confirmed by Roelofs et al. (1985), who found that the Mg, K and Ca concentrations were lower under conditions of high ammonium concentrations in the soil in *Pinus nigra* var. *maritima* stands. This, in turn, increases the imbalance of the nutrients available to the plant, changing the K/N, Mg/N, Ca/N and P/N ratios. Further research is needed, however, to reveal the role of the cation/N ratios on the infection of host by pathogens. Nutrient imbalances have been shown to affect the amount and species composition of mycorrhiza in a variety of tree species. A nitrogen fertilization of a *Picea abies* stand decreases the production of basidioma by mycorrhiza (Wiklund et al. 1995). An increased N input to oak stands, also causes a decrease in mycorrhiza as well as changes in the species variety (Zak 1964) increasing the risk for pathogen infection of the roots. Mycorrhiza is generally considered to provide protection against pathogen attacks. An increased nitrogen concentration in the soil is correlated with an increased amount of N-based compounds such as free amino acids (arginin, glutamic acids) in the tree. This has been shown for Scots pine (van Dijk and Roelofs 1988, Rabe and Lovatt 1986). Nitrogen fertilization also causes increased spring time levels of free prolin in Scots pine (Näsholm and Ericsson 1990). Volatilization of ammonia from fur farms was found to result in a 102 -103 percent increase in the arginine concentration in Scots pine needles (Pietilä et al. 1991). Nutrient deficiencies, especially of K and P, can result in elevated arginine concentrations in pine (Krupa et al. 1973, Krupa and Brännström 1974). Balsberg Pählsson (1992) presented further evidence for an increase in amino acids as a result of fertilization. The experiments demonstrated that free amino acids, glutamic acid and aspartic acid predominate in beech together with glutamine and asparagine.

Armillaria root rot

Armillaria root rot on both coniferous and deciduous trees is caused by the pathogen *Armillaria* sp. A direct positive correlation between nutrient imbalance and the occurrence of the pathogen has been shown. Experiments with pine have demonstrated that a high N deposition reduces both the mycorrhizal abundance and the quantity of the fine roots. The declining capacity of the tree for water and nutrient uptake causes a increase in drought susceptibility. Consequences are an increasing damage by *Armillaria* root rot (Ritter 1990). Entry et al. (1991) showed, by use of seedling of five coniferous tree species, that the severity of root rot was greater when the availability of light or N was limited as compared with a balanced situation. The seedlings also contained lesser amounts of phenolic compounds and higher levels of sugar in the root tissue.

Seedlings of *Pinus monticola* grown with a complete nutrient supply and full availability of light were found to have significantly lower *Armillaria* infection rates than seedlings grown under limitations of light, nitrogen or phosphorus (Entry et al. 1986). Rykowski (1983) showed, however, that a high amount of N in soils did not affect the abundance of *Armillaria mellea* in *Picea abies*.

Phytophthora root rot

During the last 20 years, the oaks (*Quercus robur* and *Quercus petraea*) in Europe have declined in vitality dramatically. The pathogen *Phytophthora* sp. has been suggested to be an important agent responsible for causing root rot in oaks (Jung and Blaschke 1995, Jung et al. 1999). Climate change and nitrogen excess may also play the role as predisposing or contributing factors for decline of this type (Jung and Blaschke 1995, Jung et al. 1996, Jung et al. 2000). More research is necessary, however, to confirm this hypothesis (Hansen and Delatour 1999, Anon. 1999).

Gremmeniella abietina/ Scleroderris canker

The growth of *Gremmeniella abietina* is favored by low temperatures (Dorwoth 1972) as well as by high moisture (Petäistö and Repo 1987). Several humid and mild winters during the last decade may naturally have promoted the occurrence of *G. abietina*. However, various investigations have shown there to be a connection between nutrient imbalance and the occurrence of *G. abietina*. An increased foliar nitrogen concentration in Scots pine stands promotes the colonization by *Gremminiella* (Ylimartio 1991). Also, the cation ratio to N seems to be of importance. A K deficiency in the needles (4.3-4.9 mg/g dw, N/K of 4.5-5.1) as well as a combination of K and Mg deficiency was found to reduce the resistance of Scots pine to *Gremmeniella*, although Mg alone had no effect on resistance to infection (Ylimartio 1991). Contrary to the these results, Ylimartio and Haansuu (1993) found in studies in vitro that an increased amount of K, increased the growth of *G. abietina*, whereas an increase in Ca negatively affected the pathogen. The same experiment showed that the nitrogen concentration is important. The Cu concentration has no effect on infection by *G. abietina* (Ranta 1994, Ranta et al. 1995).

The deposition of acid compounds seems to be both direct and indirectly correlated with the occurrence of *Gremminiella abietina*. Bragg and Manion (1984) found that a lowered pH by acid rain is related to an increase in *Gremminiella* in Red pine. Indirectly, acid mist may promote the pathogen through having a negative effect on the epiphytic microflora living on the conifer shoots. There seems to be an antagonistic correlation between the amount of endophytes and the successful infection by the pathogen (Barklund and Unestam 1988). Experiments with acid mist treatment of Norway spruce and Scots pine showed that the epiphytic microflora significantly reduce the occurrence of *G. abietina* (Ranta and Neuvonen 1994). However, other experiments with Scots pine showed no correlation to be found between sulphur deposition and the short-term colonization of *G. abietina* (Ranta and Neuvonen 1994, Ranta et al. 1995, Vuorinen and Uotila 1997). In experiments in vitro, Laflamme and Yang (1994) studied the correlation between the occurrence of *G. abietina* and of antagonistic endo- and epiphytes on *Pinus resinosa*.

The experiments revealed interactions between a number of endophytic fungus species and *G. abietina*. The results showed that 10% of the endophyte species inhibited *Gremmeniella* and that 13% parasitized on the pathogen. Nutrient imbalance and deficiencies in K, Mg or P in forest soils are known to increase the concentration of aminoacids in foliage (Rabe and Lovatt 1986, van Dijk and Roelofs 1988, Pietilä et al. 1991). High levels of amino acids significantly affect the growth of *G. abietina*. In vitro studies have revealed that a tenfold increase in arginine, glutamic acids and proline promoted myceliar growth of *G. abietina* but a hundredfold increase in arginine reduced myceliar growth (Ylimartio and Haansuu 1993).

Other pathogens and parasites

Investigations of the effects of nutrient imbalance on a number of other pathogens have been carried out. The results show that nutrient imbalance favours the presence of pathogens on the host. Bark necrosis on *Pinus nigra* var. *laricio* caused by *Sphaeropsis* increased when the tree was fertilized by ammoniumsulfate. However, a simultaneous addition of K inhibited the lesion development (De Kam et al. 1991). *Salix* sp. stands fertilized by nitrogen were more susceptible to *Melampsora* sp. than non-fertilized ones (Orians and Floyd 1997). An increased N fertilization also increased attacks by *Botrytis cinerea* and *Sacchihantes abietis* on *Picea abies* as well as attacks by *Phomopsis* sp. and *Apiognomonina errabunda* on beech (Flückinger and Braun 1998).

Conclusions

A variety of studies have shown that nutrient imbalance, caused by an increased deposition of nitrogen and/or a decrease in base cations have an important impact on attacks on trees of insects, herbivores and pathogens. The conclusions are

- A nutrient imbalance negatively affects the synthesis of carbon-based defense substances (e.g. phenolics and terpenes). This probably results in an increased attack by insects and pathogens.
- Both nitrogen fertilization and soil acidification reduce the amounts of protecting mycorrhiza on the roots, allowing antagonistic pathogens (e.g. *Armillaria* sp., *Phytophthora* sp., *Gremmeniella* sp.) to infect the tree.
- Some studies show that the increased risk of insect and pathogen attack can be counterbalanced by vitality fertilizing by nutrients, e.g. potassium. However, more research is needed.
- More research is needed to study the difference in sensibility to pests and pathogens between different species and life strategies under conditions of nutrient imbalance.

8.6 Countermeasures against stress symptoms

K. Sonesson, A. M. Jönsson, B. Nihlgård and G. Thelin

The acidification of forest soils has increased during recent decades. This has resulted in low pH-values, low concentrations of base cations, low base saturation and high concentrations of aluminium in the mineral soil. Furthermore, inorganic aluminium has increased in the soil- and runoff-water. During 1997 the National Forestry Board in Sweden proposed a program to counteract the forest soil acidification. The main aims of the program are

- to sustain the productive capacity of the soil
- to retain the vitality of the trees

- to lower the leakage of Al and other elements to the surface and ground water
- to counteract the negative effects of acidification on flora and fauna

With focus on the causes and the effects of the ongoing soil acidification a number of different countermeasures are available. The countermeasures can be divided in three different categories

- restrictions of air pollutants
- adaptation of silvicultural methods
- supply of lime, wood-ash or vitality fertilizers.

In this chapter certain of the countermeasures available are presented, recent advances in connection with mixed-species stands, liming and vitality fertilization within the SUFOR-project being focused on.

8.6.1 Restrictions of air pollutants

Of the preventive measures against soil acidification, a decrease of sulphur and nitrogen deposition is particularly critical to making long-term biogeochemical sustainable forestry possible. Considerable efforts internationally have resulted in a decrease in sulphur input, whereas the nitrogen input has remained more or less stable during the last decade (UN/ECE and EC 2000). At present, there are few indications of any substantial reductions in nitrogen deposition and this cannot be expected to occur unless governmental policies are altered. In addition, even if nitrogen deposition is decreased, countermeasures will be needed in order to restore the nutrient capital at depleted sites. Further restrictions of air pollutants are urgently needed, although this topic will not be discussed further in this chapter.

8.6.2 Adaptation of silviculture

Silviculture as such has a strong influence on the acidifying processes and the nutrient stock in forest soil. The size of the withdrawal of biomass at harvest, the choice of tree species, the method of soil scarification and the use of clear-felling or shelter-wood affect the soil in different ways. Certain measures used in forestry only affect the topsoil, whereas others can induce changes in deeper soil horizons and also in the runoff water. The use of monocultures, whole-tree harvesting, soil scarification, and the deforestation of large areas increase leaching losses and remove nutrient capital (Rosén and Lundmark 1990). The removal of nutrients per time unit is reduced if rotation periods are extended (Kimmins 1997). This is seldom an option, however, due to financial demands. The growth and increase of biomass is an acidifying process in itself. The balance within the plant between cations and anions is maintained through the roots of the plant giving away a corresponding ion when taking up a cation or an anion. Since the uptake of cations is higher than the uptake of anions, the uptake of nutrients entails a contribution of hydrogen to the soil and consequently also to the acidification process. The surplus of base cations within the plant is released when the plant dies and decomposes or is permanent taken away from the forest ecosystem at harvest. As a consequence, the soil nutrient balance is negatively affected by the harvest.

Whole-tree harvesting, or slash removal, substantially increases the loss of nutrients from a forest ecosystem as compared with stem harvesting, since branches and foliage contain large amounts of nutrients (Mälkönen 1976, Olsson et al. 1996). According to modelling of weathering rates and to calculations of mass balances, the risk of nutrient imbalance and associated negative effects on tree vitality are dramatically increased when whole-tree harvesting is practiced in a system in which N deposition and soil acidification already lead to depletion of the soil nutrient pools. After clear-felling, the leaching of nutrients will be high until a new vegetation cover has been established. Recent studies have shown there to be dramatic decreases in the amount

of leached nutrients in shelter-woods having 150 stems ha⁻¹ as compared with clear-fellings (Karlsson et al. 2000). Hence, shelter-woods can serve both natural regeneration and nutrient conservation. By using shelter-wood instead of clear-cutting, the risk of leakage of nitrate and eventually of other nutrients (e.g. potassium) decreases. Different tree species influence the soil pH and other soil properties in different ways (Raunkiaer 1922, Hesselman 1926, Nordén 1992). Spruce forest is more effective, for instance, than deciduous forest trees in collecting dry deposition. There are also differences between the various tree species in production, root depth and litter quality. A species with high production increases the acidification process due to nutrient uptake. Tree species with a deep root system can make use of a larger soil volume and can more readily redistribute nutrients within the soil horizon. On the other hand, a deep root system is not the same as an active nutrient uptake further down in the root system. Litter from some of the deciduous tree species (e.g. birch, alder, ash and elm) results in a higher pH after decomposition and a greater pool of exchangeable base cations than conifers. As a consequence, an increase in the share of deciduous trees in the forest implies a more positive nutrient situation in the long run. An increase in the proportion of deciduous trees in the forests of southern Sweden has been put forward as one of the most important measures in order to improve the possibilities of achieving sustainable management of forest resources.

Mixed-species stands

One way of increasing the share of deciduous trees is the use of mixed-species stands. Several studies show there to be higher pH and/or base saturation in mixtures of conifer and deciduous species than in conifer monocultures (Troedsson 1983, Liljelund et al. 1986, Klemmedson 1987, Frank 1994). However, little research has been done on tree nutrient status in mixtures as compared with monocultures. Increased foliage N and P concentrations and increased growth have been reported in mixtures of Sitka spruce and Scots pine or larch in Great Britain (Carlyle and Malcolm 1986, Brown 1992, Morgan et al. 1992). Total production in mixtures of conifers and deciduous trees may exceed the production achieved in monocultures (Kely 1988, Tham 1988, Debrinyuk 1990, Mård 1996, Man and Loeffers 1999), although there are many examples of the opposite occurring as well (e. g. Burkhart and Tham 1992). One reason for the higher production may be the increase in nutrient availability for one or more of the species in a mixture. The combination of fast growing Norway spruce and oak at Jämjö, in southeastern Sweden was found to result in lower nutrient losses and greater resource utilization than in Norway spruce monoculture, while simultaneously being financially more competitive (Ståål 1986). At Asa Research Park, mass balance calculations suggest that on a large part of the research park area the mixed-species stands are more biogeochemically sustainable than the Norway spruce monocultures. In order to investigate the nutrient status of Norway spruce in mixtures of deciduous species, a survey of 45 mixed-species stands on 30 sites in southern Sweden and eastern Denmark was performed (Thelin et al. 2001a). Norway spruce in mixtures containing beech, birch, or oak had higher foliage concentrations and ratios of K, P, and Zn to N than pure Norway spruce stands, Fig. 8.17. Suboptimal nutrient levels were much more frequent in monocultures. Spruce nutrition was also positively affected in stands with less than 50% deciduous basal area. This does not seem to be an effect of growth dilution, since Norway spruce growth, as well as needle N, in mixtures and monocultures were found to be comparable. However, differences in the top mineral soil between mixtures and monocultures were smaller than expected. One possible explanation for this is that soil samples were taken too close to the spruces in the mixtures (Thelin et al. 2001a). The positive effect on needle K and P status is interesting since studies have indicated declining trends of K and P in Norway spruce of southern Sweden (Aronsson 1985, Thelin et al. 1998). The spruce nutrient status in monocultures growing on fertile soils and in mixtures growing on poor soils was found to be similar. Thus, it appears that the inclusion of deciduous species in Norway spruce stands can make up for poor soil conditions. This supports the idea of using mixed-species stands to counteract nutrient imbalance in Norway spruce.

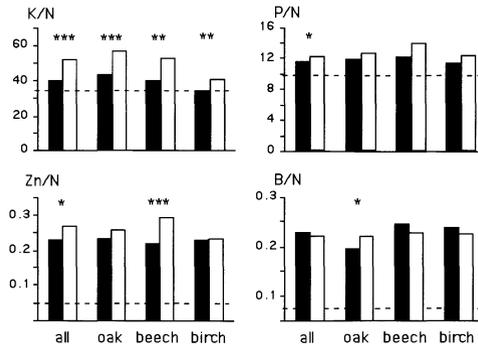


Figure 8.17: Medians of ratios of K, P, Zn, and B to N in current year needles of Norway spruce in monocultures (dark bars) and in mixtures (light bars) for oak ($n=16$), beech ($n=13$), birch ($n=16$), and all combined ($n=34$). Only nutrients for which there were significant differences compared with the monocultures are included. Dashed lines show target levels according to Linder (1995). $*=p<0.05$, $**=p<0.01$, $***=p<0.001$ (paired *t*-test or Wilcoxon signed rank test).

Improved nutrient status in Norway spruce in mixtures than in monocultures is probably the result of a combination of factors, Fig. 8.18 (Thelin, 2000)

- Litter quality is higher in mixtures of Norway spruce and deciduous species than in monocultures. Deciduous litter is generally more nutrient-rich than spruce litter (Mikkola 1985, Liljelund et al. 1986, Man and Lieffers 1999). Understorey litter, largely absent in Norway spruce monocultures, may further promote litter quality (Tappeiner and Alm 1975). The mixing of litters increases the mineralization rate and the nutrient availability (Chapman et al. 1988, Taylor and Parkinson 1988, Briones and Ineson 1996, McTiernan et al. 1997). Litter of higher quality provides a more suitable environment for soil fauna, e.g. earthworms, which in turn speeds up both decomposition and mineralization (Mikkola 1985, Saetre et al. 1999).
- If more light reaches the forest floor, at least in the spring prior to leafing, the soil temperature and thus the mineralization rates are increased (Mikkola 1985).
- The possible intraspecific rooting volume is greater in a mixture of species having different rooting patterns than in a monoculture. The total root occupation of the soil is greater in mixed-species stands than in monocultures (Debrinyuk 1990, Brown 1992, Morgan et al. 1992). This implies the total nutrient availability experienced by each individual tree to be greater. Nutrients taken up from deeper soil layers by a deep-rooted species reaches the organic layer through litterfall, thus increasing the nutrient availability for the shallow-rooted species and promoting litter quality (Man and Lieffers 1999). Roots in deeper horizons in a mixture can take up nutrients leached from superficial layers. Also, there may be a division between species regarding the timing of nutrient uptake (Chapman 1986) and the chemical forms in which nutrients are acquired.
- The canopy filtration of air pollutants is greater in Norway spruce monocultures than in mixed-species stands. Norway spruce monocultures need to buffer a larger amount of deposited acidity and take care of more deposited N than hardwood stands do (Brown and Iles 1991, Bergkvist and Folkesson 1995).
- N retention by an active understorey may lower the N availability to the trees, reducing the risk of tree-nutrient imbalance. Also, deciduous species have a greater N concentration

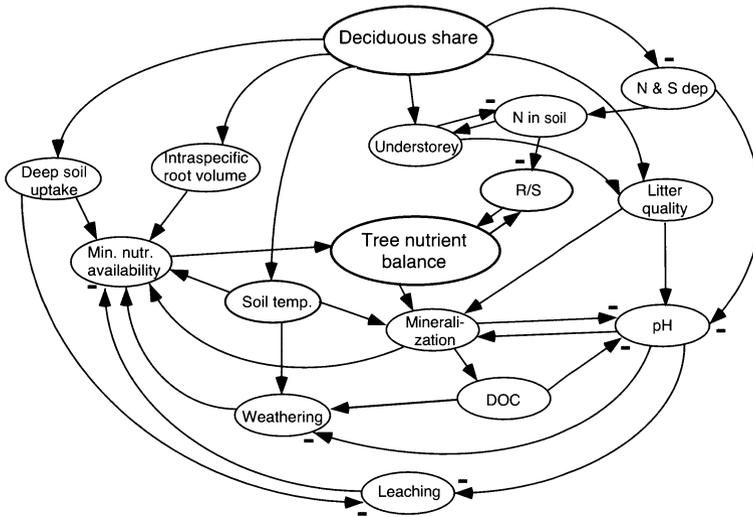


Figure 8.18: *Effects on tree nutrient balance in mixed-species stands compared with Norway spruce monocultures in an N and S deposition environment. An increase in the share of deciduous trees in the mixture affects the nutrition of Norway spruce positively. See comments in the text.*

per biomass volume than Norway spruce (Nihlgård 1972, Thelin et al. 2001a).

Several of the factors referred to above may also account for the limited nutrient losses from mixed-species stands. This is even more likely if the management of the mixed-species stands does not involve a clear-cut phase.

Growing shallow-rooted species with a high growth rate in monocultures may result in nutrient depletion of top soil layers. The lower growth rate found in a mixed-species stands and the lesser removal of nutrients from the site per unit of time would increase the possibilities of achieving tree nutrient balance. However, even if the growth rate in a mixture should equal that of a Norway spruce monoculture, the nutrient uptake is distributed within a larger soil volume thus reducing the risk of nutrient depletion (Thelin, 2000). The possibilities of achieving increased production by use of mixed-species stands instead of monocultures was already understood by forest managers in the early 19th century. Af Ström (1837) noted that site production increased in mixtures consisting of one species with a deep root system and another species with a shallow root system. It appears that differences in deposition levels and rooting depth are of considerable importance in the calculation of mass balances, which show a greater biogeochemical sustainability in mixed-species stands than in spruce monocultures. Further study of the connection between rooting depth and nutrient uptake is needed.

8.6.3 Application of lime, wood-ash and mineral fertilizers

Liming is a traditional way of counteracting forest soil acidification and increasing the soil capacity to neutralise acidifying agents. Ordinary limestone is dominated by Ca-carbonate, whereas dolomitic limestone is dominated by both Ca and Mg carbonates, the concentrations of other mineral nutrients being low (Eriksson 1993). Vitality fertilization is a nutrient and/or lime application. The main purpose of the treatment is to counteract a nutritional disorder of the forest ecosystem. Additionally, the acidification of the forest soil is prevented in the long-

term perspective (Liljelund, 1990, Nilsson and Wiklund 1995, Johansson et al. 1999). However, the concept focuses on the effects on tree nutrient status and tree vitality. Since N is not normally included in vitality fertilization, it has also been termed N-free fertilization. Applying ashes from forest residues recirculates those elements, that are taken from the forest during tree harvesting. The recirculation of wood ashes (Bramryd and Fransman, 1995) would be a natural way of returning nutrients removed at harvest. However, logistic systems for the handling of ashes are yet to be implemented and the development is restricted by the contamination of heavy metals in ashes (Bramryd et al. 1996). Application of wood-ashes decreases the soil acidity and the Al^{3+} concentration in the soil water as effectively as lime, and might be preferred, provided the content of heavy metals is kept low (Bramryd and Fransman 1995). Site-specific fertilizers with a balanced combination of mineral nutrients may often promote a rapid and sustained revitalization of trees suffering from nutrient deficiencies (Hüttl 1990b, Evers and Hüttl 1991). There is presently a need of treating about 2 million hectares of forest area in Southern Sweden (Westling et al. 2000), the size of this area increasing year by year (Kalén 1998). Especially forest soils with a pH (H_2O) below 4.5-4.7 at the depth of 20-30 cm in the mineral soil (B-horizon) are considered to be in immediate need of treatment (Westling et al. 2000). To reach the goal of affecting the runoff water, forest soil treatment should be carried out over an entire watershed (Westling et al. 2000). The National Forestry Board in Sweden recommends treatment by dolomitic lime of 3 ton ha^{-1} or a mixture of wood ash and dolomitic lime (2 + 2 ton ha^{-1}). Liming in doses of 3-4 ton ha^{-1} prevents an increase in soil acidification, and has a positive influence during a period of 30-40 years without having any negative side effects (Staaf et al. 1996). At higher doses, an increased nitrification may occur, that might cause a slight nutritional imbalance, and the number of fine roots and roots infected by mycorrhiza may be adversely affected (Andersson and Persson 1988). The supplying of lime, wood-ash or mineral fertilizers may increase the leaching of base cations, whereas the concentration of aluminium and hydrogen in the run-off water usually decreases. On forest soils with a low ratio of C to N (eg. $C/N < 25$), an increased nitrification and level of NO_3 in the soil water may appear. In a growing stand, however, the ability of trees to take up the released nitrate is usually sufficient to take charge of the available NO_3 . A high availability or excess of Ca after liming can reduce the tree uptake of K (Tomlinson 1991, Ljungström and Nihlgård 1995). The effect of the supplying of lime or vitality fertilizers on soil chemistry, leakage, soil processes and organisms depends on the type of material used, the size of the granules, the dosage and of the properties of the soil that is treated. Adding moderate doses that have a low rate of dissolution reduces the risk of nitrate leaching. Results from three early Swedish experiments, Långban, Venjan and Dalby that were started in 1907, 1913 and 1951 respectively, indicated clearly that liming has long-lasting positive effects on the pH value, exchangeable base cations, base saturation, the lowering of Al and plant species diversity (Nihlgård and Popovic 1984, Nihlgård 1996). After only a few years, the soil effects are most evident in the topsoil, whereas the mineral soil at 20-30 cm depth is first affected after a period of 10-30 years (Andersson and Persson 1988).

The environmental impact assessment carried out for the countermeasure program of the Swedish Forestry Board (Johansson et al. 1999) concluded that up to 1997 no evidence had been found that the production of forest stands had been lowered due to soil acidification, the growth rate of the forests in Sweden generally continuing instead to increase. However, if soil acidification continues, as has been shown by repeated measurements in Sweden (Kalén 1998), the production capacity will very likely decrease. Simultaneously, much evidence has appeared showing soil acidification to be linked to a negative development of flora and fauna and water acidification to an increasing Al-content in the water. A calculation of the socio-economic costs (Kalén 1998) concluded that liming of acid forest soils in Sweden was economically worthwhile, provided that treatment mitigated a future growth decline of two to five percent and assuming that the effect lasted for 20-40 years. There is a clear risk that future costs will be many times as high as the present costs of mitigation. To improve the sustainability and vitality of the entire forest ecosystem, different kinds of countermeasures are under investigation. The positive effects of vitality fertilization on tree nutrient status and tree vitality have been demonstrated in many

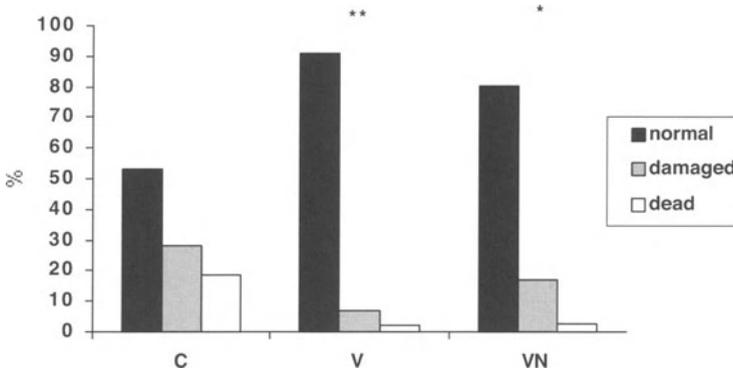


Figure 8.19: Changes in classes of crown defoliation from 1994 to 1998 in two oak stands treated with 1 ton Skogvital PK 4-7 Mikro ha⁻¹ in 1994 (n=146).

European studies (Hüttl 1990a,b, Katzensteiner et al. 1992, Boxman, et al. 1994, Flückiger and Braun 1995, Katzensteiner et al. 1995). Until recently, a positive response in terms of growth and vitality after treatment by N-free fertilizers in Sweden had not been documented (Nohrstedt 1990, Nohrstedt et al. 1993, Johansson et al. 1999). The lack of response in such experiments is probably due to that the trees did not suffer from a nutrient deficiency. If nutrient deficient stands had been used, as most often has been the case in European studies, it is likely that stronger effects would have been observed. An increased growth or an improved vitality following treatment should not be expected after the addition of non-deficient nutrients.

Within SUFOR, research has focused on restoring nutrient imbalance in trees and the effects on tree vitality, such as on sensitivity to frost, degree of crown defoliation and crown structure. Different kinds of countermeasures have been investigated and site-specific adaptations of treatments have been considered. It is important to evaluate soil and tree chemistry prior to treatment in order to avoid negative effects and maximize treatment efficiency. Also, the form in which the nutrients are added, e. g. granulated or in combination with organic material, as well as the type of application, e. g. aerial or to the ground, may be important to treatment effects. Recent SUFOR results on liming and vitality fertilization are

- Beech trees treated with lime showed an increase in the Ca/Al ratio and 50% less Mn in the bark tissue as compared with the control trees, indicating changes in nutrient uptake due to higher Ca supply and less acid soil condition (Jönsson 2000 a). A tendency toward fewer bark lesions was found on trees seven years after liming with 5 ton ha⁻¹ (Jönsson 2000b). Ca stabilises the cell walls, and high levels of Ca may therefore reduce the severity of bark necrosis (Perrin and Garbaye 1984). Although bark lesions on beech are often caused by frost damage (Day and Peace 1934) a test of frost resistance failed to show any significant difference between trees treated with lime and untreated control trees (Jönsson 2000b).
- Three oak stands in southern Sweden were in 1994 treated with 1 ton Skog-Vital PK 4-7 Micro, a nitrogen-free mineral fertilizer. Two of the oak stands responded as early as one or two years afterwards with an increased foliar mineral content of B, K, Mn and P, Tab.8.11. After four growing seasons, the crown development in two of the stands was more positive ($p < 0,05$) in the treated plots than in the control plots, Fig. 8.19, whereas oaks on the treated plots showed a tendency towards lower stem diameter growth than in the control plots (Sonesson 2000).
- A declining Scots pine/Norway spruce stand in central Sweden with indications of B and Cu deficiency was treated with a site-adapted fertilizer in 1996. After one growing season,

| | | Trälshult | | Hjälmseryd | | Älmta | |
|---------------------|------|----------------|------------------|----------------|----------------|----------------|-------------------|
| | | Control | Treated | Control | Treated | Control | Treated |
| B, $\mu\text{g/g}$ | 1994 | 36.2 \pm 8.3 | 38.3 \pm 9.5 | 20.7 \pm 5.1 | 15.4 \pm 8.0 | 18.7 \pm 5.0 | 18.6 \pm 4.4 |
| | 1995 | 32.8 \pm 7.5 | 37.0 \pm 6.2 | 18.4 \pm 9.2 | 22.6 \pm 6.1 | 26.5 \pm 7.6 | 40.2 \pm 9.5** |
| | 1996 | 17.3 \pm 4.8 | 20.7 \pm 7.5 | 9.4 \pm 4.8 | 12.5 \pm 5.2 | 14.8 \pm 3.9 | 23.2 \pm 4.0*** |
| K, $\mu\text{g/g}$ | 1994 | 9.4 \pm 0.9 | 9.2 \pm 1.4 | 9.7 \pm 2.3 | 9.0 \pm 1.8 | 13.1 \pm 1.5 | 10.3 \pm 2.0*** |
| | 1995 | 7.7 \pm 1.4 | 8.4 \pm 1.2 | 7.2 \pm 1.5 | 6.9 \pm 1.0 | 8.5 \pm 1.4 | 8.0 \pm 1.5 |
| | 1996 | 5.8 \pm 0.9 | 7.0 \pm 0.7** | 6.3 \pm 1.2 | 6.4 \pm 1.6 | 9.6 \pm 2.6 | 8.6 \pm 2.2 |
| Mn, $\mu\text{g/g}$ | 1994 | 1.8 \pm 0.6 | 2.1 \pm 0.5 | 1.2 \pm 0.4 | 0.9 \pm 0.3 | 1.8 \pm 0.5 | 1.9 \pm 0.2 |
| | 1995 | 1.9 \pm 0.3 | 2.1 \pm 0.4 | 1.1 \pm 0.4 | 1.3 \pm 0.5 | 1.9 \pm 0.4 | 2.4 \pm 0.3** |
| | 1996 | 1.9 \pm 0.2 | 2.4 \pm 0.3** | 0.7 \pm 0.7 | 1.1 \pm 0.9 | 1.5 \pm 0.9 | 2.6 \pm 0.6** |
| P, $\mu\text{g/g}$ | 1994 | 1.7 \pm 0.2 | 1.9 \pm 0.3 | 1.4 \pm 0.2 | 1.3 \pm 0.1 | 2.4 \pm 0.3 | 2.2 \pm 0.2 |
| | 1995 | 1.6 \pm 0.2 | 1.9 \pm 0.2*** | 1.2 \pm 0.1 | 1.2 \pm 0.2 | 2.1 \pm 0.3 | 1.8 \pm 0.2* |
| | 1996 | 1.6 \pm 0.2 | 2.0 \pm 0.2*** | 0.6 \pm 0.7 | 0.9 \pm 0.6 | 1.8 \pm 0.2 | 1.9 \pm 0.2 |

Table 8.11: *Foliar nutrient concentrations and standard deviations of three Swedish oak stands treated by 1 t Skog-Vital PK 4-7 Micro 1994-1996 (n=54).*

the trees showed increased needle B and Cu concentrations and increased needle mass and dbh (diameter at breast height). The response in dbh remained for five years following treatment. Two years after treatment, the treated plots showed a reduced proportion of trees with needle loss >30%, less discoloration of needles, improved growth of top shoots and improved vitality of leader shoots (Thelin and Nihlgård, 200x).

- In a greenhouse experiment, beech seedlings were grown in acid forest soil with and without the addition of an organic fertilizer consisting of decomposed grape seeds. Treated seedlings responded with increased growth, decreased root/shoot ratios, and an increased shoot concentrations of several nutrients, the response being strongest for Cu, K and B. This implies there to be nutrient deficiencies in seedlings grown in unfertilized soil (Thelin and Nihlgård, 1998). The nutrient loading of seedlings in nurseries through application of organic fertilizers prior to plantation might be a possible routine to increase plant survival and establishment following plantation in the field.

8.7 Forest conditions — A regional overview

Peter Schlyter, Kerstin Sonesson, Ingrid Stjernquist, Gunnar Thelin

8.7.1 Background

That the occurrence of extensive forest damage symptoms in southern Sweden, as in the rest of Europe, in the early 1980ies represented a largely unexpected situation is underlined by the fact that the then current official forest monitoring systems initially were, by and large, inadequate to provided useful data on damage levels and distribution, nor needed background information on what could be regarded as normal damage levels.

The official Swedish National Forest Inventory (NFI) provided no defoliation or needle loss data prior to 1984. However, the NFI was designed to be a strategic resource information system providing statistical data on national forest resources and for the evaluation and development of national forest policy. Data collection was based on a rigorous statistical design and collected

over a five year campaign period. It should be recognised that the NFI was designed to monitor rather gradual changes in parameters like age, species composition, timber volumes for the country as a whole or for larger survey regions and not originally organised to monitor rapid changes or fluctuations, between years, in the Swedish forests, and certainly nor to provide early warning of forest health. When the need to collect forest vitality/decline data, i.e. data on defoliation and foliage discolouration, became apparent the NFI was the obvious institutional platform for this. Basically it was a matter of adding some further parameters to be recorded by the field personnel, though in practice it raised questions of choice of sample size, criteria as well as standardisation and training issues. However, even when this type of data started to be collected one could argue that the official monitoring system was still somewhat lacking in spatial and temporal resolution, owing to NFI original design, and in particular in perspective of reports of a dramatic rise in forest damage in the Federal Republic of Germany where the proportion of reported damage on older forest went from about 10 to 50% these shortcomings to a large extent still remain unresolved. Other types of forest decline indicators than defoliation and discoloration like foliage chemistry have been slow in introduction within the NFI framework.

8.7.2 Regional co-operation for assessment of decline and stress

Owing to an early concern about forest decline in Scania, southern Sweden, and an identified urgent need for data to assess the perceived threat to forest vitality an ad hoc group against forest damage (Samrådsgruppen mot skogsskador) for collaboration between the regional Forestry Commissions (Skogsvårdsstyrelsen), the County Environmental Agencies (Länsstyrelsens miljövårdsenhet), Lund University scientists and forestry sector representatives was established in 1984. The group initiated monitoring work along several lines. It established a more dense network of permanent monitoring plots supplementary to the official Forestry Commission scheme where e.g. needle loss assessment was performed bi-yearly, and needle and soil chemistry were sampled; but also initiated pilot studies on high resolution regional forest decline surveys based on photo-interpretation of large scale colour-infrared aerial photographs and carried these to operationalisation. Several of the approaches pioneered by the group have subsequently become adopted for regional surveys or incorporated in national or other regional monitoring programmes.

8.7.3 Regional forest decline surveys

The degree of defoliation is the basis for most operational forest decline surveys, in Sweden as in Europe. Defoliation has, however, been criticised as a vitality/decline variable on the grounds that it is unspecific, i.e. that a tree may suffer defoliation for a number of reasons some natural other anthropogenic. One could on the other hand argue that its lack of specificity is its advantage in a survey context where one wants to assess the combined impact of multiple stress factors - rather than using a variable as a proxy for a specific environmental load. Comparing defoliation with other measures of vitality for Norway Spruce Salemaa and Jukola-Sulonen (1990) concluded that defoliation may be regarded as an integrated measure of tree vitality. The usefulness of defoliation as a vitality or productivity measure has been questioned on a more fundamental level e.g. by Innes (1993a) who claim that there is no clear relationship between defoliation and production. Nevertheless, studies from France (Becker 1990), North America (Barnard et al. 1990) and Sweden (Söderberg 1993) point to a relationship between defoliation and production. Söderberg demonstrated, in a major Swedish study on Norway Spruce (n=16,650) and Scots Pine (n=15,600), a strong statistical relationship between per cent stem growth (based on coring, normalised to fully foliated trees at the sample site) and estimated needle loss. Similarly, schemes based on defoliation, changes in apical growth or branching pattern have also been used to assess decline in deciduous trees. An other approach for monitoring vitality is represented by chemical analysis of foliage and root bioassays.

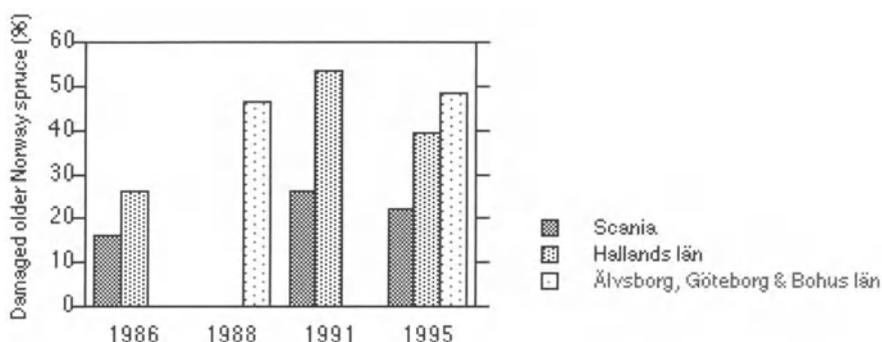


Figure 8.20: Forest decline development for older Norway spruce in Scania, Halland and Älvsborg, Göteborg and Bohus län regions (Schlyter and Anderson 1997).

Norway Spruce and Scots Pine

High resolution regional forest decline surveys of Norway Spruce (*Picea abies*) and Scots Pine (*Pinus silvestris*) have been carried out in various parts of southern Sweden since the mid 1980ies (Schlyter and Persson 1986; Schlyter 1987a,b; Schlyter and Andersson 1992; Schlyter and Anderson 1997) but the methodology have also been used in other parts of the country. The methodology is based on stereoscopic photo interpretation of defoliation using large scale (1:2,000) colour infra-red aerial photographs (CIR). The methodology provides a much larger sample than the NFI thus allowing regional and sub-regional decline assessment; i.e. the NFI provide about 0.5 and 0.3 Norway Spruce and Scots Pine per 1,000 hectares of forest whereas the aerial surveys provide 87 and 19, respectively. Forested land was air-photo sampled in a 5 by 5 km grid and approximately 1,500 sample points were photographed between mid July to the end of August, once the beginning of September. The photographs were taken with Williamson F-49 Mk 3 cameras with 30 cm focal length lenses equipped with Wratten 12 filters on 23x23 cm Kodak 2443 Aerochrome film. Stereoscopic photointerpretation was performed by specially trained personnel from the various regional Forestry Comissions using either Zeiss Jena Interpretoscopes or Leitz Aviopret 2 equipment allowing variable magnification of the photos. At each sample point species, age class, defoliation of individual trees were assessed within 16 sub-sample plots, with a total area of 1 hectare. The subsamples were defined centred within the stereo model on a clear photographic film. The defoliation assessment was made in 20 per cent classes and the individual plots were registered with respect to altitude, aspect and position with respect to nearness to exposed stand margins; tree top breakage and Peridermium damage was also assessed. The accuracy of the defoliation assessment was checked by subsequent field controls as well as in separate methodological studies. Defoliation assessment by photo interpretation is as accurate, or slightly better than, conventional needle loss assessment made in the field by trained personell of the NFI (Schlyter and Bengtsson 1990; Schlyter and Wulf 1997). For the operational survey 1995 the deviation of a group of nine photo-interpreters from the independent field control assessment of % varied between 0.0 - -2.9 for Norway Spruce (s.d 1.4 - 6.1) and 1.4 - -3.6 (s.d. 0.3 - 3.5) for Scots pine. The forest decline between 1986 to 1995 can be assessed for several regions in southern Sweden (Schlyter and Anderson 1997). However, owing to a dependency on local Forestry Commission initiative and funding constraints full coordination of efforts were not achieved until 1995. In 1995 the counties of Scania, Halland, Blekinge, Älvsborg, Göteborg and Bohuslän as well as parts of the counties Kronoberg and Jönköping.

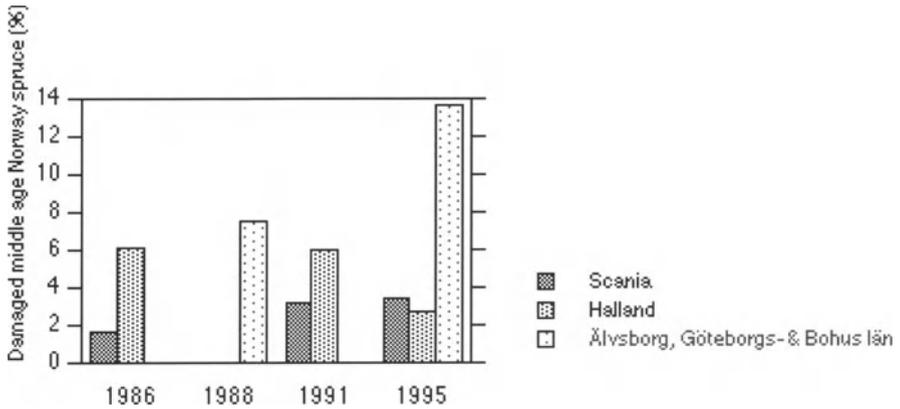


Figure 8.21: Forest decline development for middle age Norway spruce in Scania, Halland and Älvsborg, Göteborg and Bohus län regions (Schlyter and Anderson 1997).

Thus the counties of Scania and Halland were surveyed 1986, 1991 and 1995, the counties of Älvsborg, Göteborg and Bohuslän in 1988 and 1995. In the 1995 survey area 38.9% of the older (defined as as above 50 yr or 60 yr depending on region) spruce and 24.7% of older pine had a defoliation above 20% i.e. were defined as damaged, cf. Table 8.12. Of the damaged trees most are to be found in the defoliation class 21-40% and proportionally fewer in the higher damage classes, and older trees are more prone to damage. An increase in the number of damaged older Norway Spruce is evident if one compares 1980ies and 1990ies survey results, Fig. 8.20, though damage levels appears to have stabilised, albeit at a higher level, in the earlier half of the 1990ies. The damage levels for the West coast counties around 40% gives clearly cause for concern. The proportion of damaged Scots Pine show a much more marked variability between years. This reflects the fact that a fully foliated Swedish pine has a maximum of three years needles as compared to seven for spruce and the loss, or partial loss, of one years needles will have a proportionally larger impact on pine foliation. The proportion of damaged spruce and pine in the middle age range (30 to 50/60 yr) show much more variation, as can be seen in Fig. 8.21 and 8.23. In spite of the apparent improvement in Scania and Halland 1995 as compared to 1991, damage levels are still higher than at the beginning of the 1980ies when the "new forest damages" were causing concern. The increased trend is also present in



Figure 8.25: % beech and oak in different defoliation classes 1988, 1993 and 1999. % distribution of beech in branching classes in 1988, 1993 and 1999. After Anderson and Sonesson (2000).

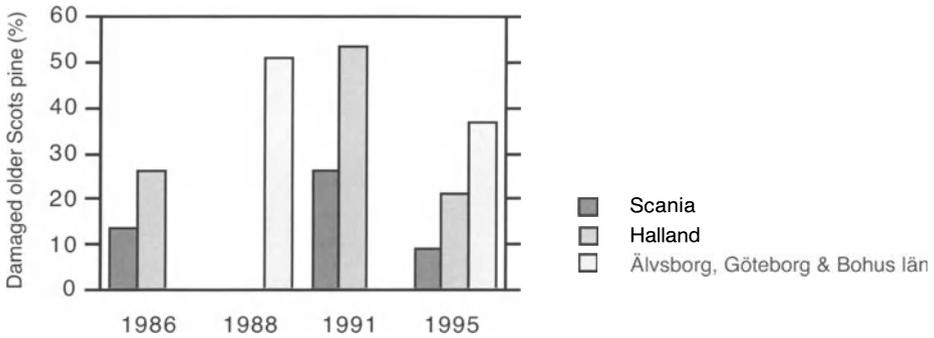


Figure 8.22: Forest decline development for older Scots pine in Scania, Halland and Älvsborg, Göteborg and Bohus län regions (Schlyter and Anderson 1997)

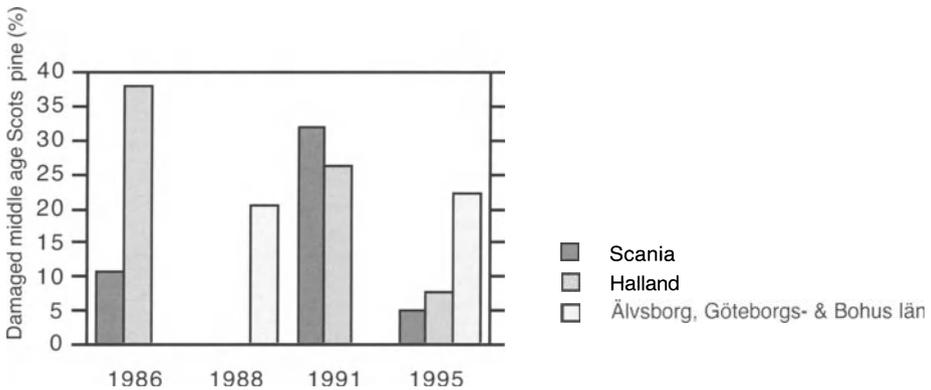


Figure 8.23: Forest decline development for middle age Scots pine in Scania, Halland and Älvsborg, Göteborg and Bohus län regions (Schlyter and Anderson 1997).

the NFI database. Decline, measured both as per cent trees with damage or as mean defoliation, broadly viewed show a similar spatial distribution both for older spruce and pine. The proportion of damaged older Spruce is presented in Fig. 8.24. Scania, and particularly the southern parts of the province has lower proportion of damage spruce. This could reflect better soil conditions but also the fact that, broadly speaking, the age of the older spruce in this area is likely to be lower than in the rest of the surveyed area. The damage concentration in Blekinge is interesting in that it is located in the vicinity of an oil fired reserve power plant with historically significant sulfur emissions. The repeated surveys also point to a complication in assessing forest vitality - namely the between years varying impact of forestry activities. Some of the most damage areas in northern Halland 1986 appeared to fare better in 1991 in spite of the rather drastic increase in spruce damage between the years a fact interpreted to reflect increases awareness of forest damage and subsequent hygienic fellings by forest owners. Improved conditions as revealed by decline surveys need not per se mean imply that the vitality of the forest has improved as a consequence of reduced stress - it could also, at least partly, be an effect of forest management. This points to the fact that even long term monitoring of defoliation, even when based on standardised methodology, have to be interpreted with some caution as changes

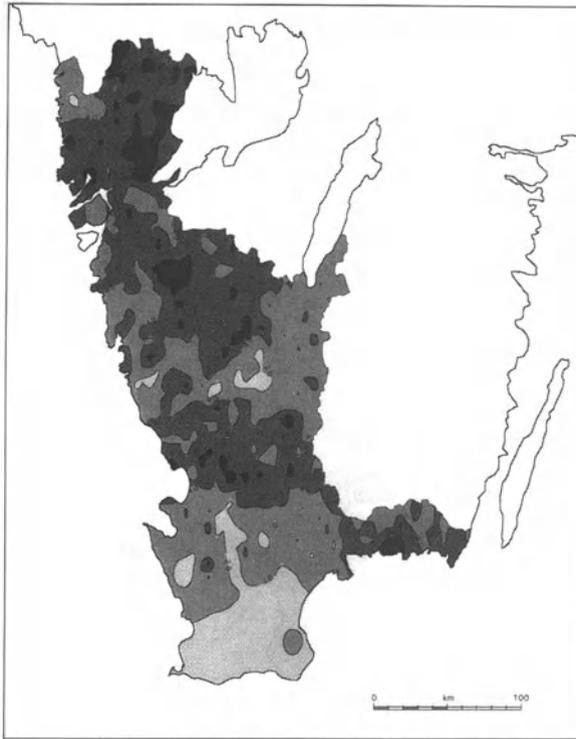


Figure 8.24: The proportion damaged older Norway spruce, i.e. trees with a defoliation >20%, in 1995. (Schlyter and Anderson 1997).

may also reflect changes in the response of the forester and not necessarily in biologic response only. From a monitoring perspective this underlines the importance to operate with several supplementary monitoring techniques, e.g. foliage chemistry, root bioassay, soil chemistry etc.

Beech and oak

The condition of beech (*Fagus sylvatica*) and oak (mainly Pedunculate oak *Quercus robur* L., and to a lesser extent sessile oak *Quercus petraea* (Matt.) Liebl.) in southern Sweden has been monitored since the late 1980s using a standardised methodology based on field observations (Wijk 1989). The monitoring network is based on a 5 by 5 km cell grid. Up to 10 observation plots were chosen along the centre and diagonals of the cell. Observation plots in forest stands of less than 1 ha, and plots without oak or beech were excluded. The remaining plots were visited, sample trees being selected according to a standardised format, cf. Sonesson (1998, 1999a,b). The field assessment was made during a six-week period, from mid-July to the end of August, by personnel of the regional Forestry Commission. Altogether nine persons were involved in the field assessment. Crown defoliation was assessed on the uppermost two-thirds of the crown and was given as a percentage of a normally foliated crown (i.e. with a 0-10% transparency). The estimated defoliation included leaf loss, crown transparency, branch loss and dying branches. Normal self-pruning by self-shading was not included. Soil samples were taken in conjunction

| | yr | Defoliation class | | | | | Mean Mean | n n |
|---------------|-----------|-------------------|-------|-------|-------|---------|--------------|--------|
| | | 0-20 | 21-40 | 41-60 | 61-80 | 81-100% | | |
| | | % | % | % | % | % | | |
| Norway spruce | >50/60 | 61.1 | 31.5 | 5.9 | 0.8 | 0.7 | 19.7 | 116355 |
| Norway spruce | 21 -50/60 | 91.7 | 7.0 | 0.8 | 0.1 | 0.5 | 12.2 | 223803 |
| Norway spruce | 1-20 | 99.6 | 0.2 | 0.0 | 0.0 | 0.2 | 10.2 | 76232 |
| Scots pine | >50/60 | 75.3 | 21.4 | 2.2 | 0.4 | 0.8 | 16.1 | 68440 |
| Scots pine | 21 -50/60 | 85.5 | 13.1 | 0.8 | 0.2 | 0.3 | 13.3 | 52361 |
| Scots pine | 1-20 | 99.0 | 0.9 | 0.0 | 0.0 | 0.0 | 10.2 | 6444 |

Table 8.12: *The per cent Norway Spruce and Scots Pine in different defoliation classes, mean defoliation (calculated on class mid value) and the number of trees classified in the forest decline survey of southwestern Sweden 1995.*

with the survey conducted in 1999 and based on 5-8 sub-samples per survey plot (cf. Andersson and Sonesson 2000). In total, 258 soil samples were taken. Decline surveys were performed in 1988, 1993 and 1999. The majority of the sites and trees were investigated all three years. The 1999 survey covered a total of 268 stands and approximately 4,200 trees, 2,606 beeches and 1,642 oaks being observed.

Beech and oak vitality in southern Sweden have deteriorated continuously since the first decline survey in 1988 (Anderson and Sonesson 2000). The mean crown defoliation for beech has increased from 5.7% in 1988, to 12.5% in 1993 and to 20.8% in 1999. Oak defoliation has increased as well, from 10.9%, to 17.4% to 33.2% in 1988, 1993 and 1999, respectively. A strong recruitment to higher defoliation classes is evident for both species, but most pronounced for oak. In 1999, 27% of the beech and 59% of the oak trees had a defoliation in excess of 25%, as compared to only 3% of the beeches and 9% of the oaks in 1988, Figure 8.25. A similar trend is identifiable in the branching pattern for beech, Figure 8.25. The picture of a rapid decline in vitality is further affirmed by assessment of the current year shoot growth of the different years. Whereas only 16% of the beeches and 4% of the oaks had a good growth in 1999, close to 60% of the trees of both species, showed a good yearly shoot growth in 1988, Figure 8.25. A regional trend among beech trees is evident from the data, with increasing incidence of damage towards the western part of the area that was surveyed. The regional trend for oak is less clear. It is interesting to note that the degree of defoliation, both for beech and oak, is correlated with the base saturation percentage. Beech stands growing on sites with a very low base saturation percentage show a significantly higher ($p=0.016$) defoliation than stands growing on sites with a normal base saturation percentage, as well as on podsols as compared with cambisols ($p=0.020$). Similarly, oak defoliation is larger in stands growing on sites with very low base saturation percentage ($p=0.026$) and on podsols ($p=0.085$) than stands growing on sites with a normal base saturation percentage and on cambisols (Anderson and Sonesson 2000).

8.7.4 Surveys of foliage chemistry

Forest growth in boreal ecosystems is normally limited by nitrogen availability (Tamm 1991, Gundersen and Bashkin 1994). In recent decades, forest productivity has increased enhanced N-deposition having been shown to be an important underlying factor (Eriksson and Johansson 1993). Increased productivity has in turn led to an increased demand of mineral nutrients by the trees, which may, depending on edaphic conditions, be growth-limiting (Nihlgård 1985, Thelin 2000). Together with nutrient loss due to the leaching of K, Ca, and Mg following soil acidification, the increased demand for mineral nutrients may result in nutrient deficiencies in the

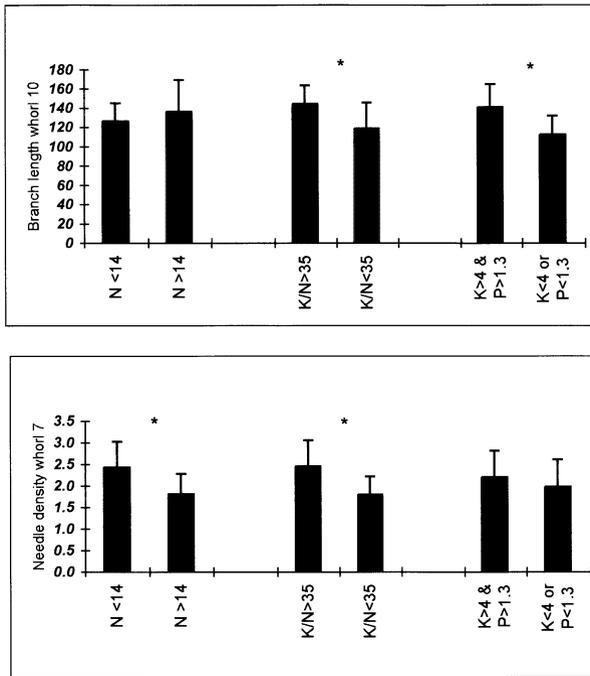


Figure 8.26: Branch length on whorl 10 (cm) and needle density (needle mass (g) / branch length (cm)) on whorl 7 in trees from stands older than 40 years in 1999 divided into groups depending on needle N (mg/g), K/N (indicated *).

trees (Hallbäck and Tamm 1986a,b; Falkengren-Grerup et al. 1987, Tamm and Hallbäck 1988). Deficiencies related to N-deposition have been shown for P (Mohren et al. 1986), K (Thelin et al. 1998), Mg and Ca (Hüttl 1990b), Ca and Mg (Katzensteiner et al. 1992), and P and Mg (Houdijk and Roelofs 1993). Depending on soil properties, the nutrient for which the supply becomes insufficient differs between stands. Aronsson (1985) found decreasing P concentrations concomitant with increasing N concentrations in Norway spruce needles during a 20 year period in southwest Sweden. Foliage chemistry - i.e. measured concentrations of K, Ca, Mg, P, Zn, Cu and N as well as ratios of these to N - have been used as vitality indicators for spruce and pine, or rather the existence of nutrient limited conditions for growth, as compared with identified threshold values.

Regional nutrient stress - Monocultures of Norway spruce

The monitoring of forest conditions in Scania, initiated in 1984 by the ad hoc Group against Forest Damage (Samrådsgruppen mot Skogsskador), has provided more than 15 years data on growth, crown condition, and tree and soil nutrition in 33 Norway spruce and 9 Scots pine stands. The database affords unique possibilities for drawing conclusions regarding the sustainability of Norway spruce monocultures southern part of Sweden. Changes in tree and soil nutrition from 1985 to 1994 have been summarized by Thelin et al. (1998b), crown defoliation during the same period was analyzed by Rosengren-Brinck et al. (1998), and changes in soil acidification until 1999 in Jönsson, U et al. (2001). Data from these plots have also been used for modeling the soil chemistry as affected by acid deposition (Barkman and Sverdrup 1996). Plots have

been utilized for studying the demand of the tree roots for N, P, and K (Rosengren-Brinck et al. 1997); the relation between N-deposition, forest floor C/N-ratios, and tree and soil nutrition (Schlyter et al. 2001); the connection between tree nutrient status and tree vitality and growth (Thelin et al. 2001b); the effects of vitality fertilization (Jönsson 2000b,c,d); and the elemental transport from minerals by the ectomycorrhizal mycelium (Hagerberg and Wallander 2001).

Random trees from each monitoring site were used for needle sampling. Current-year needles from the first or second order side branches on the seventh branch whorl from the top were sampled during the winter season (Dec-March). Five trees were used in 1985, and six trees in 1987, 1990 and 1994. In order to minimise the effects of individual variability, the same trees were used for needle sampling the various years. Soil samples were taken as 15 subsamples along each diagonal of the sample plot and were fused layer-for-layer, yielding two samples per plot and layer. For further details and a description of the analytical procedures, see Thelin et al. (1998). Scanian Norway spruce and Scots pine stands have developed a relative K deficiency (Thelin et al. 1998). By 1994 a majority of the monitored stands had developed needle K/N-ratios below the target level of 35 (Linder 1995). The negative trend in needle K-status could not be explained in terms of increasing tree age or of climatic cond

The existence of K deficiency was further supported by root bioassays performed on material from 12 of the Norway spruce stands that indicated K stress (Rosengren-Brinck et al. 1997). During recent years, from 1988 to 1993, the Ca, K, and Mg concentrations in the top mineral soil have decreased, but increased in the B-horizon, indicating there to be a leaching of base cations from superficial to deeper soil layers (Thelin et al. 1998). This was accompanied by a decrease between 1993 and 1999 in the base cation concentrations deeper in the mineral soil (Jönsson, U. et al. 2001). Today, the vast majority of plots show base cation levels in the mineral soil that are below what is considered necessary in order to maintain a productive, sustainable forestry in the long run. Of the plots, 58% have a base saturation below 5%, a level considered extremely low for a productive forest soil, (Jönsson, U. et al. 2001).

The link between deposition of acidifying compounds and N and declining tree and soil K status in Scania is illustrated by the co-occurrence of areas with high deposition and low needle K status, Fig. 8.27. In addition, computer modeling of the effects of deposition on forests in Scania support the assumption that decreasing K-concentrations and K/N-ratios may be explained by deposition of N and S. Barkman and Sverdrup (1996) used soil data from the 33 Norway spruce plots together with deposition data, Fig. 8.27, and climatic data. Mass balances for the base cations were calculated, using weathering rates calculated by PROFILE (Warfvinge and Sverdrup 1992). The results indicated that in a long-term perspective there is a high risk of K becoming deficient in the soil solution. The model prediction is validated by the extremely low levels of exchangeable K in the mineral soil in 1999 (Table 8.13, Jönsson, U. et al. 2001) and the development of K deficiency in the trees (Thelin et al. 1998). The C/N-ratio in the organic layer has been suggested to be a suitable indicator of the risk of nitrate leaching (Gundersen et al. 1998) and consequently also of the development of nitrogen saturation. The C/N-ratio is not the cause of leaching, but rather is a symptom of imbalance between nitrogen immobilization and carbon sequestration in the soil. In 1999, the C/N-ratio in nearly half of the Scanian Norway spruce stands was below 25, indicating there to be a high risk for nitrate leaching (Table 8.13). In addition, there has been found a positive correlation between needle K status and C/N-ratio in the humus layer, indicating there to be a negative relationship between

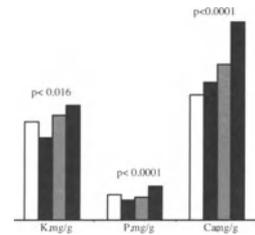


Figure 8.28: Nutrient concentrations in beech leaves from Maglehem in SE Scania, June, July, August, September 1990, growing on Dystric cambisol. a) Nitrogen and potassium, b) Phosphorus and magnesium (Tyler et al. 1992).

| Parameter | Ref. value | Year | Mean ⁴ | % |
|------------------------|------------------|------|-------------------|------|
| Al, µg/g | 100 ² | 1988 | 129.5 | 61.3 |
| | | 1993 | 188.7 | 80.6 |
| | | 1999 | 200.5 | 83.9 |
| pH(BaCl ₂) | 4.2 ² | 1988 | 4.27 | 32.3 |
| | | 1993 | 4.07 | 71.0 |
| | | 1999 | 4.10 | 71.0 |
| Ca, µg/g | 50 ² | 1988 | 119.5 | 78.1 |
| | | 1993 | 116.6 | 71.9 |
| | | 1999 | 88.4 | 81.3 |
| K, µg/g | 20 ² | 1988 | 13.3 | 75.0 |
| | | 1993 | 17.6 | 62.5 |
| | | 1999 | 11.6 | 100 |
| Mg, µg/g | 10 ² | 1988 | 7.2 | 78.1 |
| | | 1993 | 7.9 | 78.1 |
| | | 1999 | 5.2 | 93.8 |
| Base sat.(%) | 20 ² | 1988 | 21.9 | 77.4 |
| | | 1993 | 20.0 | 80.6 |
| | | 1999 | 13.4 | 83.9 |
| C/N ¹ (%) | 25 ³ | 1993 | 26.5 | 38.7 |
| | | 1999 | 25.9 | 45.2 |

Table 8.13: Means and percentage of plots with soil chemical values below (pH, base cations, base saturation and C/N) or above (Al) the suggested minimum or maximum levels for the exchangeable phase at 20-30 cm mineral soil depth in a productive long-term sustainable boreal coniferous or mixed forest soil (Jönsson, U. et al. 2001). % = % of plots deviating from reference values. 1; Organic layer, 2; see Forest health indicators, 3; Gundersen et al. (1998), 4; median for pH.

N availability and tree nutrient balance in Norway spruce stands in Scania.

The nutrient imbalance, manifested by low needle K and K/N and high K demand, may be seen as the first step in the decline in tree vitality. However, crown defoliation studies generally show tree age to be the most important factor in explaining needle loss (Rosengren-Brinck et al. 1998), although there are significant negative correlations between needle K and P status and needle loss as well. However, in stands younger than 40 years the branch length was positively correlated with needle N status, although in older stands this relationship could not be found (Thelin et al. 2001a). Rather, in older stands branch length was correlated with needle K status. This indicates a movement from N-limited growth in young stands (<40 yrs) to K-limited in older stands. In addition, shoot multiplication rate appeared to be positively correlated with needle K status but not with needle N status or with tree age (Thelin et al. 2001b). Older stands (>40 yrs) with needle N concentrations of less than 14 µg/g had a higher needle density (needle mass/branch length) than stands in which needle N was above 14. Stands in which needle K/N was above 35 had higher needle density and branch length than stands in which needle K/N was below 35, Fig. 8.26. Hence, in stands older than 40 years nutrient imbalance appears to affect tree vitality negatively. The lack of similar effects in younger stands may be due to the high N demand owing to high growth rates in younger stands. Young trees seem to withstand a suboptimal mineral nutrient levels better. Younger trees may possibly be better in translocating K to where the demand is greatest, which with increasing organism size may become increasingly difficult. In conclusion, the vast amount of collected and analyzed data on

| | Beech | | Oak | |
|----------|-------------|-----------|-------------|-----------|
| | Bergmann | Class 2 | Bergmann | Class 2 |
| N, mg/g | 19 -25 | 18 -25 | 20 -30 | 15 -25 |
| Ca, mg/g | 3.0 -15 | 4.0 -8.0 | 3.0 -15 | 3.0 -8.0 |
| K, mg/g | 10 -15 | 5.0 -10.0 | 10 -15 | 5.0 -10.0 |
| Mg, mg/g | 1.5 -3.0 | 1.0 -1.5 | 1.5 -3.0 | 1.0 -2.5 |
| P, mg/g | 1.5 -3.0 | 1.0 -1.7 | 1.5 -3.0 | 1.0 -1.8 |
| Cu, µg/g | 0.035 -0.1 | - | 0.035 -0.1 | - |
| Zn, µg/g | 0.015 -0.05 | - | 0.015 -0.05 | - |
| Ca/N | 16-60 | 22 -32 | 15 -50 | 20 -32 |
| K/N | 53 -60 | 27 -40 | 50 | 33 -40 |
| Mg/N | 7.9 -12 | 5.5 -6.0 | 7.5 -10 | 6.7 -10 |
| P/N | 7.9 -12 | 5.5 -6.8 | 7.5 -10 | 6.7 -7.2 |
| Cu/N | 0.03 -0.05 | - | 0.03 -0.04 | - |
| Zn/N | 0.08 -0.2 | - | 0.08 -0.17 | - |

Table 8.14: *Optimum foliar nutrient concentrations and target ratios to N for beech and oak. Bergmann: values defined by Bergman (1988). Class 2: Classification values for class 2 according to the 3rd Forest Foliar Expert Panel Meeting (Anon. 1997b).*

the Scanian Norway spruce stands reveals a picture of progressing negative effects of acid and eutrophicating deposition on soil acidity and nutrition, on tree nutrition, and on tree vitality and growth. Growing Norway spruce in monocultures located in a high-deposition environment, such as in southern Sweden, does not appear to be sustainable. Today, chemical changes in Scanian soils have developed to a stage of which countermeasures are needed in order to restore the tree nutrient balance and tree vitality. Nutrient addition experiments in a limited number of stands have shown positive effects on frost sensitivity and on needle and bark nutrition, there being indications too of decreased resin flow. Against this background further nutrient addition experiments are clearly needed.

Target values for beech and oak foliage chemistry in Sweden

In order to use foliar nutrient concentrations, or nutrient ratios to N, to assess vitality of deciduous tree species, target values for the most essential nutrients need to be defined for each species. The suggestion by Linder (1995) that all tree species have the same optimal nutrient status requirements implies there to be no need to identify specific target values for deciduous trees. Given the variation in growth strategies, in adaptation to different edaphic and successional stages this appears to be a bold assumption. One may, perhaps, group together deciduous tree species with similar growth strategies and assign them common target values. On the other hand, since the boundary between nemoral and boreal forests passes through southern Scandinavia and many of the broad-leaved tree species have their northern limit in southern Sweden, a difference in nutrient requirement for Swedish provenances compared to European ones would appear likely. In the Scandinavian context forest decline studies have concentrated on the economically important species and identification of threshold values for late successional species such as oak and beech are urgently needed. Those for earlier successional species such as birch are important, but are probably less urgent from an environmental perspective.

In the classical work of Bergmann (1988) concerning the nutrient conditions of cultural plants, including forest trees, the optimum nutrient levels given for beech and oak are well

| | Kongalund July 1967 | Öved Oct 1967 | Linderöd Oct 1967 |
|----------|------------------------|------------------|----------------------|
| N, mg/g | 25 | 27 | 21 |
| Ca, mg/g | 5.0 | 8.8 | 6.1 |
| K, mg/g | 5.8 | 7.7 | 4.9 |
| Mg, mg/g | 1.7 | 2.0 | 1.4 |
| P, mg/g | 1.6 | 1.1 | 1.9 |
| Ca/N | 20 | 32 | 29 |
| K/N | 23 | 28 | 23 |
| Mg/N | 6.8 | 7.5 | 6.7 |
| P/N | 6.2 | 4 | 8.7 |

Table 8.15: *Foliar nutrient concentrations and nutrient ratios to N for mature beech stands in Scania, Sweden, 1967 (Nihlgård and Lindgren 1977)*

above the values for conifers, Table 8.14. To identify concentration spans, as well as ratios to N, Bergmann used all European information available at that time. The drawback of this approach was that he pooled nutrient values, independent of the sampling time of the year, the age of the individual tree and of location. The information originated mainly from young plants growing in experimental gardens, its general validity for mature trees growing under natural conditions by no means being obvious. A similar problem regarding differences in age and time is apparent in the compilation of literature data by van den Burg (1985 and 1990). An additional problem in this dataset is the various and unharmonized methods of sampling and analysis. Since quite different terms were used for the same values or categories of values in Europe (Bergmann 1988, van den Burg 1985, 1990), a new classification, class 1-3, was decided upon by participants in the meeting of the 3rd Expert Panel in Vienna in 1995 in order to avoid misinterpretation, Table 8.14. A new database for nutrient concentrations in tree foliage in Europe have been established within the framework of ICP Forest and the European Union Scheme on the Protection of Forests against Atmospheric Pollution (Anon. 1997b). To identify the natural nutrient status of Scandinavian beech and oak, information from the period prior to that of the high deposition of nitrogen or from areas outside the influence by air pollutant deposition would be needed. Since beech only forms forest stands in the southern part of Sweden, no sites outside the impact of high nitrogen input can be found. Only three early reliable data sets on foliar nutrient status are available. All of these were collected from beech stands in Scania, growing on podzol (Linderöd) and on cambisols (Kongalund and Öved), respectively, Table 8.16. Compared with the optimum nutrient values given by Bergmann, the Ca, K, Mg and P concentrations at the Swedish sites are close to or below deficiency. However, for two of the stands the foliage was collected in October, i.e. at senescence, when the trees had started to retranslocate N and P (Staaf and Stjernquist 1986). The Kongalund data was collected in July, when beech leaves often show a dip in nutrient concentrations. No data on oak foliar nutrient concentration in Sweden prior to 1990 is available. The foliar nutrient concentrations of deciduous tree species change during the vegetation period, depending on leaf growth and seed development. In monitoring forest foliar chemistry, leaf samples are most appropriately collected in August since leaf development is finished then and the autumn retranslocation has not yet started (Tyler et al. 1992).

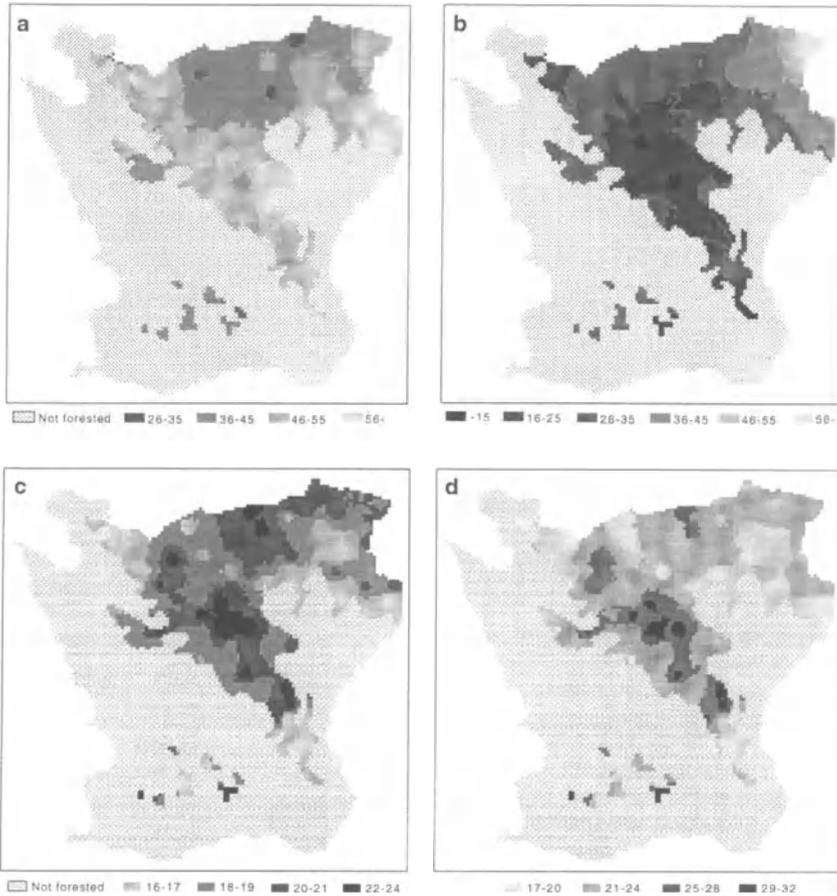


Figure 8.27: Interpolated maps of K/N (%) in Norway spruce needles in Scania in 1985 (a) and in 1994 (b) (adapted from Thelin et al. 1998). Total deposition (kg ha⁻¹yr⁻¹) of N (c) and S (d) in the forested parts of the study area in 1988 (adapted from Barkman and Sverdrup 1996).

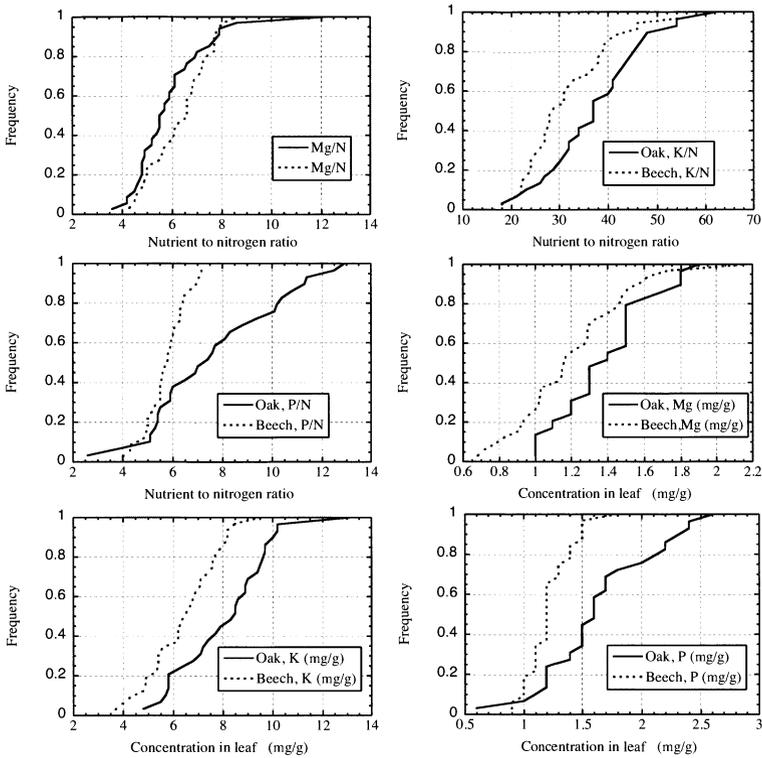


Figure 8.29: Frequency graph of the foliar concentrations (bottom) and foilar ratios (top) of K, Mg, and P to N in mature beech and oak trees in southern Sweden during 1985-2000.

The current foliar nutrient status of mature beech and oak stands in southern Sweden was investigated by evaluating all existing information since 1985, Fig. 8.29. The beech dataset covers the last 15 years and the oak set the period since 1990. The Forestry Commission as well as various research projects have been responsible for foliar sampling and analysis. The dataset includes stands from varying soil conditions, ranging from cleyey till to poor till soils. However, the sampling technique and the nutrient analysis methods are identical for all investigations. The average K, P and Mg concentrations in Sweden are well below the European levels given by Bergmann (1988). For beech, the concentrations are close to the lowest values in the ICP forest span for Europe, cf. Table 8.14. For both oak and beech, the average foliar Zn concentrations are higher in Sweden than in the studies given by Bergmann. The K, P, Mg and Cu ratios to N for Sweden are all below the figures given by Bergmann. Compared with the ICP forest dataset, Swedish oak and beech trees have average Mg to N ratios that are at the lower end of the European span. This is also true for the P/N ratio in beech. The beech sites in southern Sweden with the lowest K, P and Mg concentrations show values and ratios to N that are below the European span. For oak, the foliar concentrations in Sweden correspond to the lowest European values, but the ratios to N are below the minimum European values reported. The major part of the foliar samples in the Swedish beech and oak investigations was taken from Swedish provenances growing on podsolis and dystric cambisols. This reflects the rather poor soil conditions for most of the existing deciduous forest stands. The deficiency levels and target values for the ratios to N may thus be lower than the main European thresholds. A first attempt

| | Beech | Oak |
|----------|-------|------|
| K, mg/g | <4.8 | <5.8 |
| Mg, mg/g | 0.8 | 1.1 |
| P, mg/g | 0.9 | 1.2 |
| Ca, mg/g | 4.0 | 4.7 |
| K/N | 22 | 26 |
| Mg/N | 4.8 | 5.0 |
| P/N | 4.8 | 5.3 |
| Ca/N | 20 | 22 |

Table 8.16: *Suggested deficiency levels and target ratios to N for Swedish beech and oak stands*

to define deficiency levels and target values for beech and oak is summarized in Table 8.16. The concentration figures given are the 10% frequency values shown in Fig. 8.29.

Discussion

The forest vitality monitoring in southern Sweden that has been carried out for more than a decade generates several questions, some of them scientific and others of a more applied character. We have already touched upon the former. Regarding the latter, one can note that both the regional high resolution decline monitoring and the spatial high resolution monitoring of foliage chemistry has been carried out on an ad hoc basis, its not being part of any long-term program ensured either economic or institutional stability. This can be regarded as rather complacent, given the fact that the forests in the southern third of Sweden are heavily impacted on by acidifying compounds, nitrogen deposition and tropospheric ozone and that these forests represent a natural resource that is responsible for roughly half the forest growth in Sweden. Forest vitality and production in this area is clearly of national interest from an environmental, economic and employment standpoint. Current monitoring, organised within the national or the EU framework, needs to be supplemented by other data. The regional monitoring that have been carried out thus far has been the result of largely regional initiatives from by Forestry Commissions and by researchers - an approach that clearly is far too dependent on local initiative and ad hoc funding to be sustainable in a long-term perspective.

8.8 Estimation of total risk for forest health

The assessment of the total risk for an ecosystem is best made by proper treatment of individual risks in a statistical methodology. The problem is complicated by the fact that even if certain types of risks can be described by symmetrical random distributions, others such as soil acidification effects, ozone, and fumigation are characterized by asymmetric and biased distributions. Risk assessments are only at an early stage of development and systems for handling acidification (Barkman 1998), sulphur dioxide fumigation (Barkman and Ardö 1999), frost and wind (Blennow 2000) still operate singly. The integration of these remain as a very large task. The difficulty rests with the intricate system of risk feedbacks within the system. Thus, the risk for frost is not dependent on low temperatures alone, it is also dependent on the trees history of ozone exposure, nutrient and water stress. Without the use of an integrated model, the assessments obviously become an exercise in gambling. Thus, with an integrated model now simply on the drawing board, our total risk estimates remain largely qualitative.

- The effects of soil acidification are amplified by acidification

| Species | Drought tolerance | Shadows tolerance | Nitrogen effect | Frost | Wind resist | Soil | Nutrient capture | Root rot | Animals browse |
|----------|-------------------|-------------------|-----------------|--------|-------------|-------------|------------------|----------|----------------|
| Spruce | poor | good | small | good | poor | moraine | 0-0.35 m | much | some |
| Pine | good | modest | tiny | good | modest | moraine | 0-0.5 m | some | much |
| Larch | poor | poor | small | modest | solid | loose | 0-0.7 m | much | much |
| V. Birch | modest | poor | modest | poor | modest | moraine | 0-0.6 m | some | much |
| G. Birch | poor | poor | modest | good | solid | loose | 0-0.7 m | some | much |
| Alder | poor | poor | small | poor | modest | wet | 0-0.4 m | much | much |
| Aspen | poor | poor | large | poor | modest | rich | 0-0.5 m | much | much |
| Cherry | poor | poor | large | poor | solid | rich | 0-0.7 m | some | much |
| Oak | modest | poor | small | poor | solid | heavy, rich | 0-1.0 m | some | little |
| Beech | poor | modest | small | poor | modest | drained | 0-0.8 m | some | some |
| Ash | poor | poor | large | poor | solid | rich, wet | 0-1.0 m | no | much |

Table 8.17: *Overview of the effects of various factors that can affect the health and vitality of different tree species, and indications for how these tend to react.*

- The effects of drought amplify the effects of acidification
- Acidification-induced root damage increases the risk of windfalls
- Drought decreases the effects of ozone
- Nitrogen availability increases the risk of frost damages.
- An increased nitrogen availability above what is essential increases the risk of fungal infections and insect attacks

Important too is the integration of environmental risks with risks of windthrow as well as with combinations of root rot, effects of soil acidity on the root system and windthrow. Windthrow risks and frost risks depend only in part on the species (Kimmins 1997) and to a large degree as well on topography.

Chapter 9

Biodiversity in sustainable forestry

Mats G. E. Svensson, Ingrid Stjernquist, Peter Schlyter, and Harald Sverdrup

9.1 Introduction

The global and regional decline in biodiversity during modern times is the result of several interacting factors due to changes in human numbers, life styles, ways of land-use and production and the associated load of environmental pollution. Current risks of irreversible loss of habitats and extinction of species is incomparably greatest in species rich tropical areas of the developing South. Nevertheless, concerns for loss of local biodiversity also figure, rightly, high on the agenda in developed North. Viewed from a Swedish perspective many of the species on the national red list are rare owing to the fact that they occur at or close to their climatically controlled distribution limits or owing to changes in land use and methods in agriculture (most are associated with the agricultural landscape) and forestry and few of these species are at risk for extinction in a wider European or global perspective (Sjöberg 1994).

The conditions for biodiversity changes with time are in principle given by current and passed climate, land use and land use technologies, geochemical conditions as well as the stock and resource status of a system in constant change (Fig. 9.1). The concept of critical load can also be applied to biodiversity in a general way, to determine the maximum amount of pollutant input that will not cause unwanted changes. This can also be extended to management, to determine the maximum amount of human impact that will not cause unwanted changes. The critical load concept can be applied to the individual components of biodiversity, such as individual species occurrence, ecological system functions or the geophysical and biogeochemical processes that support and affect the ecological system. Preservation of the present biodiversity status is less important than the preservation of the structural complexity and convergent ecosystem dynamics. Present biodiversity status is in all cases a result of the climate, biogeographical and biogeochemical history coupled with the impacts of man on the landscape. It is a result of a long management history, a pollution history as well as the ecological development of the landscape with different types of impacts. The establishment of relevant endpoints and landscape management goal states will be of importance for biodiversity management, and the endpoints are in themselves moving targets. Future climate scenarios for the Nordic countries presented by The Swedish Regional Climate Modelling Programme predict an increase in yearly average air temperature of 2.5-6°C and a significantly wetter climate in the west by the year 2100. All predictions available to us for the next 200 years, suggest that human physical impact on the landscape will continue, and that both climate and environmental pollution are likely to change almost dramatically. Thus, we must realize that biodiversity of the Swedish landscape will remain in a state of change for the foreseeable future. We cannot change the fact that it will change, and our environmental goals must incorporate the concept of change towards ecologically desirable endpoints.

Our interpretation is that the biodiversity refers to the different kinds of plants, animals and other living organisms in all their forms and levels of organization that exist in an area. It includes:

Genetic diversity – The distribution of genetic variation occurring in a particular population of a species.

Species diversity – The number of species inhabiting a certain area and the amount of each species in that area.

Ecosystem diversity – The number of different species inhabiting a particular area, and relates to the variety of habitats, biotic communities, and ecological structural processes, and the variety of ecological processes (alpha diversity).

Landscape diversity – The number of different species inhabiting different ecosystems in a particular geographical region, and relates to the diversity of ecosystem types in that area, their geographical extent, geographical connectedness and population stability robustness. (beta diversity)

Preserving biodiversity and practicing sustainable forestry are globally stated aims in several international accords, above all in the CBD (European Commission, 1995). Consequently, the issue of integrating environmental values and multiple benefits of forests into practical forest management has gained wide interest. Biodiversity is one of the most difficult environmental problems to predict and describe in a way that allows design, planning of mitigation strategies. Biodiversity management has often focused efforts on saving high-profile endangered species. The problem with this species-by-species approach is that it is expensive and often unsuccessful when there are no habitats left for the endangered species, it generally ignores the ecosystem perspective, though a focus on one species have in some cases resulted in an ecosystem approach to its preservation. An alternative approach to regional biodiversity management has been to develop a more or less comprehensive inventories of the status of specific taxa, indicators or species in a region, and through information systems, negotiation and land planning, to develop management strategies. It is a problem that the approach focus on quantity of parts and components, but seldom on the fact that interactions of the parts in the system may be superior in importance to the number of parts or value coefficients attached to specific components. On an operational level such surveys are unlikely, except for limited areas, to complete in a spatial

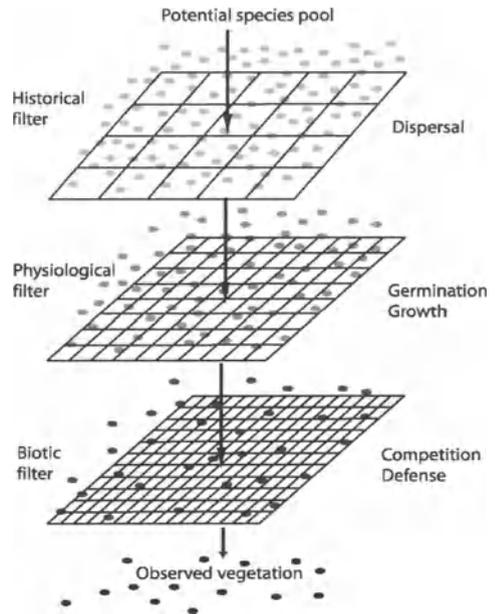


Figure 9.1: *The difference between potential biodiversity and observed is explained by the action of several "filters". Every area is suitable for a certain array of species. The history of the area, and the dispersal capacity of species in neighbouring area decides which species that will be present in the potential species pool; the historical filter. Physical conditions and physiological aspects will decide which species in the potential species pool that will germinate and/or establish itself; the physiological or abiotic filter. The availability of habitable sites, competition (within and between species), predation and herbivory will then decide which species that will actually survive and propagate; the biotic filter.*

sense and unlikely to cover more than taxa of interest to the interested amateur biologists i.e ornithologists, flora project groups etc. The usefulness for biodiversity planning even of complete datasets but with limited spatial (e.g. 5 by 5 km) resolution is not to be taken for granted (Johansson 2000) even though some limitations may be overcome by remote sensing, GIS modelling and GAP analysis. Biodiversity loss links to almost all general environmental problems. Experts give biodiversity loss rank seventh as a priority problem, and public opinion ranks it eighth (European Commission, 1995). Available evidence on several species shows a declining trend of diversity within ecosystems, habitats and among species in the EU (EEA, 1996). Climate change, acidifying compounds and nutrient nitrogen are expected to have individually and integrated significant effects on the biodiversity. Landscape history is important determinant for the resource stock at hand, as well as the structural properties of the landscape ecosystem. There will always be a difference between the natural potential biodiversity and the biodiversity actually observed.

Biodiversity is always impacted beyond the natural, because all landscapes inhabited by man are to some degree cultural landscapes. The important question is how to define the sustainable biodiversity. This may be seen in a critical load context as what is the minimum acceptable and sustainable biodiversity in a whole landscape unit. This calls for methods of expressing biodiversity in quantitative and measurable terms. The threats to biodiversity are several, and in modern landscapes, several will be relevant at any time:

- Direct species loss of small scattered populations, caused by high vulnerability to minor changes in habitat and basic conditions. Hope of spontaneous recovery is virtually nil
- Habitat loss and fragmentation of the landscape caused by agriculture, urbanization, industrialization, communication infrastructures and other economic activities
- Change in disturbance regimes (hydrology, fires, large scale vegetation changes, forest stand structure caused by management, coastal processes)
- Change the physical and chemical conditions of the ecosystems owing to pollution
- Climate change causing change the geographical location of current habitats for various species (Sykes et al 1996, Cowling et al 2001). A threat potentially aggravated by habitat discontinuity and a potentially fast rate of change in climate

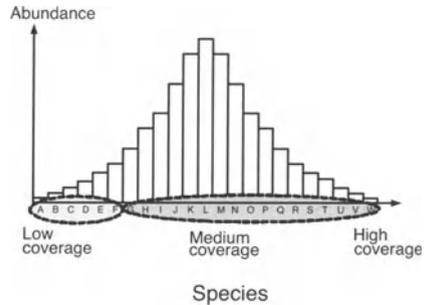


Figure 9.2: *In assessments of effects, both the rare and the common is important. There will be a number of species with low abundance, but high area coverage, which will contribute much to the character of the landscape and the bulk of the vegetation, whereas other rare species will be low in both abundance and area coverage. In SUFOR both aspects are important for the assessment of effects of management, climate change and pollution change.*

9.2 The Swedish cultural landscape and biodiversity

The gradual change in landscape from a "natural state" to one governed by the needs of human habitation and associated activities for population suspension (agriculture, forestry, industry, infrastructure) is, however, a much more long-term process reflecting the steady increase in the human population over millennia. The change from hunting and gathering to agriculture and the subsequently intensified use of the landscape for agriculture transformed, indeed transplanted, most European forest ecosystems already in pre-history (Atkins et al. 1998).

Remaining forest and woodland were in varying degrees used or managed for human needs. Larger more pristine forests survived longer in less populated areas e.g. northern Scandinavia. In Europe to day few and minuscule forest areas can be regarded as pristine (Peterken 1996).

The general structure of the cultural landscape in southern Sweden was established already by the Late Iron Age and persisted well into the 18th century (Widgren 1983). As such the land use and ecosystems reflected the limits posed by geologic conditions, climate, available technology and human pressure, Fig. 9.1 By the 18th century human population increase and use of forests in southern Sweden had transformed them considerably, the extent of the forest was reduced, as was its age and forests were fairly open in character. Forest had been cleared for permanent fields for farming, to provide areas for haymaking but also by rotational farming after burning on a couple of decades return period. Forests were integrated in the in-field and out-field agricultural production system. Forests were usually grazed, often pollarded closer to villages, and apart from grazing provided fuel, material for crafts and construction timber. Population increase increased the pressure on the agricultural system.

Signs of a non-sustainability, with a loss of meadows to crop production and an increased use of forests for grazing and fire-wood, were at hand in many areas of southern Sweden by the early 18th century (Anderberg 1991, Gadd 2000). As a consequence of these pressures forests were by 19th century rather open, low in timber, relatively young and large areas in the south and south-western part of the country had been transformed into treeless Calluna heaths used for grazing. Further increases in population resulted in a tide of crofters settling in forests outside villages adding to the exploitation of the forest resource through additional grazing and small scale clearances for crop production in direct association with the croft. Drainage projects, an increase of agricultural acreage, settlement in the northern part of Sweden, emigration to North America all helped to ease an even harder use of the land resources. The introduction of fertilizers and modern agriculture transformed farming productivity thereby reducing land use pressures on the forests, and the last century has seen yet another major change in the forests in southern Sweden. Reforestation programmes targeting the man made heaths, government forest policies and the establishment of a strong Forest Service organisation, support for forest drainage, road building, programmes to exchange mixed or broad-leaved forests on former grazed land with spruce mono-cultures etc have given the result that forests in southern Sweden are more closed, older and contain larger timber volumes than they have for centuries. From a biodiversity point of view we may note that forest cover a larger proportion of the landscape than they have in historic times (Emanuelsson in SNV 1993) have a smaller proportion of broad-leaved tree species, less bog and less bog forests. Modern forests in southern Sweden contain little dead wood (but not necessarily less so than during previous periods of intense use as grazing lands and a fuel resource). Indeed, larger amounts of old broad-leaved trees (e.g. *Fraginus*, *Ulmus*, *Tilia*, *Quercus*) with dead wood were more associated with pollarding and animal husbandry than with forests proper. The only larger stable broad-leaved forests were composed of beech. Lately, new landscape scale changes in biodiversity, e.g. an increase in thin-leaved grasses and an accompanying decrease in Blueberry (*Vaccinium myrtillus*, owing to systemic biogeochemical change through the deposition of trans-boundary air-pollution have been observed (Rosen et al. 1992).

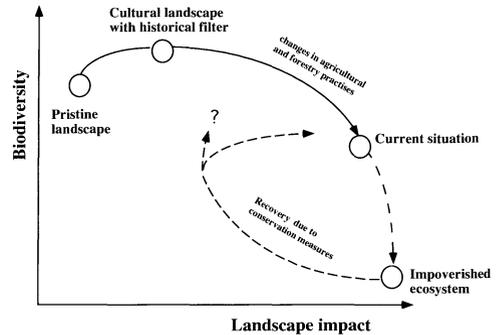


Figure 9.3: *Ecosystems are initially resilient to change, but after reaching the critical load of impact, a large change can be expected. In biodiversity, recovery does not follow the same path and may involve long delays. The societal implication is that prevention is economically superior over restoration.*

9.3 The need for system analysis of biodiversity

An operational framework is needed for analyzing terrestrial ecological systems from local to national scales and, in so doing, aims to define the role of GIS and environmental modelling have to play in the conservation of biodiversity as reported by the European Environmental Agency (EEA) (see www.biodiversity.be/bbpf/econf/econfdath.html for statements). It is an accepted principle that appropriately scaled information is required to underpin the complex decision making needed to understand, maintain and restore biodiversity. Ongoing research in the SUFOR programme and other landscape ecological research has shown that landscape history is very important for biodiversity. The changes caused by climate and humans over the centuries are still evident in the landscape structure and species pools. This stresses the importance of a long-time perspective when analyzing the state of the present biodiversity in an ecosystem as well as the chosen end point for restoration.

Agriculture has continuously changed the landscape of southern Sweden during a period of 6,000 years (Welinder et al 1998). However, the impact was never evenly distributed among ecosystems, the farming and forest managing methods of a certain time period decided which kind of ecosystems was most affected. For example the first farmers were dependent on the high nutrient concentrations in the soils of the broad-leaved forest, changing the soil characteristics and exchange the forest stand from dominated by broad-leaved tree species to birch, hazel and alder. On the other hand, forest and agriculture land are never isolated from other activities in the society. Industrialisation and urbanisation has changed, often irreversible, the habitats of many species.

During the last centuries this influence has grown in space and time, from the local to the global, from short term easily reversible change to slow long-term changes difficult to identify and reverse (Lundgren 1994). The impact may also be substantial before any significant response in biodiversity can be identified, due to the resistance and resilience of the ecosystem. An ecosystem may possess several stable states and the predisposing effect of one or multiple impact factors working over a certain time period suddenly push the ecosystem to a new visible state (May 1977, Larsen 1995). Therefore, the change of biodiversity over time is not linearly. A loss of resilience can even result in a catastrophic shift into a more impoverished state (Scheffer et al 2001). The following factors may be of importance for loss of resilience:

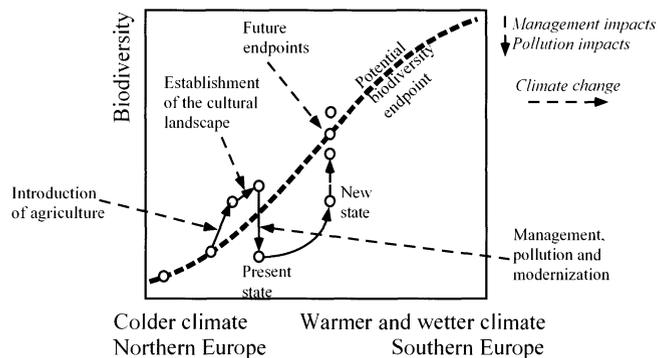


Figure 9.4: *Climate and pollution change will change the endpoints for growth as well as biodiversity in the next 200 years. The drawn line represents the maximum potential of growth or biodiversity as a function of climate change towards wetter and warmer, roughly corresponding to a north-south gradient in Europe. Significantly more growth than now will be possible, and it is a challenge for the future to be able to exploit this for profits in a sustainable way in the future. Biodiversity will never again become what it once was, nor will we keep what we have now, instead we will move towards new endpoints. The standard preservation and conservation paradigm may no longer be valid under such circumstances, because the very endpoints may change in a rather fundamental way. Any biodiversity conservation policy must consider this in its biodiversity strategy.*

- Regional climate change
- Specific forest ecosystem or forest resources exploited by man
- Forest management methods
- Alteration of biogeochemical cycles

Like many environmental problems the time delay in identifying changes in biodiversity depend on the difficulties to interpret early warning signals and the ecological impacts as well as decision making concerning effective countermeasures. Fig. 9.3 and 9.4 show the long-term historical change in biodiversity caused by various impacts on the landscape. The biodiversity state of an ecosystem may show significant hysteresis in recovery, associated with very long delays in response. It must also be remembered that some biodiversity changes are more or less irreversible, and once effect has occurred and manifested itself, there is no return to the initial state. The distinction of when the effect of an impact is reversible or irreversible is quite uncertain and a precautionary principle may be warranted. The loss of biodiversity describes both the dimension of change and the speed of the process. Compared to a historical and more sustainable use of the forests, the current biodiversity state is a result of an accelerated loss of species caused by the anthropogenic stress factors as well as the forest management methods of the last decades. Future threats like climate change and an increased need of forest resources for society purposes, i.e. bioenergy, may push the ecosystem to an even more impoverished biodiversity state. Another important, and probably the most important threat is the potential drastic decrease in agriculture land that will turn many Swedish regions into a landscape dominated by forests. The recovery process may involve going back to conditions with substantially less human impact than was required to initiate the change. The implication for society is that it will be much more expensive to restore a damaged ecosystem biodiversity than to prevent the damage to occur. To maintain or restore biodiversity is not a question of creating a forest ecosystem that can hold as many species as possible. As discussed above to maintain the biological processes and functions of the ecosystem is a prerequisite for species survival. A restoration decision must hold in mind that the resilience of the ecosystem and its key organisms may change according to landscape fragmentation as well as the filter effects caused by an increased competition capacity of subordinate vegetation, pests and herbivores (Grime 2001). Only a system analysis of these factors can identify possible end-point ecosystems of recovery or restoration. On the other hand many discussions of biodiversity restoration identify the pristine forest ecosystem, i.e. the situation before any significant human impact, as a suitable end-point state. However, the characteristics of the pristine ecosystem is hard to identify due to the historical filter described in Fig 9.1. It may also be impossible to reach if the climate and ecosystem processes has irreversible changed over time, even if the forest stand will be free from further human impact. If the pristine ecosystem can be described, restoration beyond the historical filter cannot be reached without heavy costs, both economic and social. With system analysis, the future development of forest biodiversity may be evaluated if the following characteristics are identified:

- the end-point ecosystem
- the long-term resilience capacity
- future environmental impacts on ecosystem resilience

Studies in the SUFOR programme will in cooperation with other research programmes, work towards this goal.

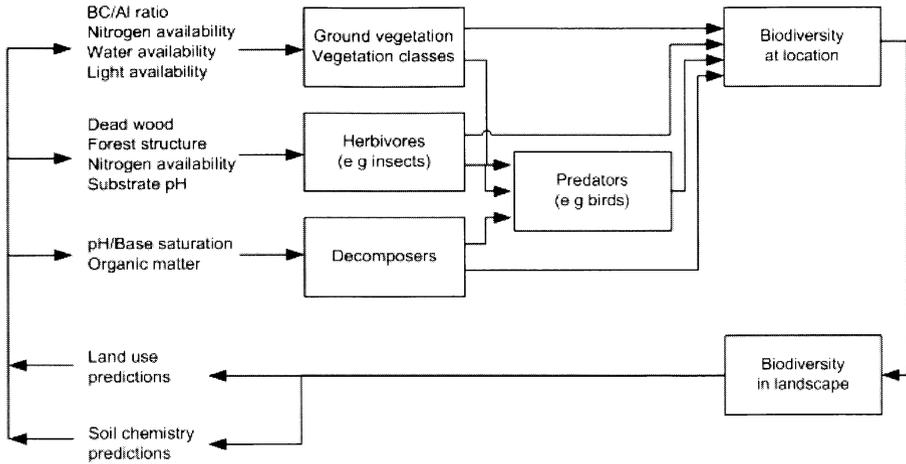


Figure 9.5: Overview of the proposed structure for a biodiversity model system.

9.4 Modelling strategy in SUFOR

The research into biodiversity in SUFOR has a very deliberate purpose of developing assessment and planning tools for practical application to management questions and problems, with the purpose to design landscape management. This implies that we want to be able to make quantitative predictions of biodiversity changes caused by forest management and other human impacts. Overview of the proposed structure for a biodiversity model system is shown in Fig. 9.5. It is important to realize that the response time to change is very different for the different boxes. Ground vegetation plants and trees are relatively mobile with respect to large scale geographical changes and may respond by moving their occurrence area. In contrast, insects as particular family of species are very much less mobile despite their motorical mobility, they are less plastic in their adaptability and can only survive under very narrow ranges of conditions. This demands a larger biodiversity and speciation rate, in order to match the larger natural variation as a genera, but leads to a larger extinction rate on species basis. Animals like birds or herbivores are also very mobile and much more adaptive, and thus a lower biodiversity is required together with a lower speciation rate. Thus the geographical adaptation time to a fundamental change is in the order of years and decades for large animals, several decades to a few centuries for plants and trees, but millennia to millions of years for insects. The numerical species count based biodiversity for the different groups cannot be linearly added without a proper transformation. A larger numerical species biodiversity in one group is not necessarily more worth than a smaller numerical species biodiversity among another group. The questions pertinent are:

- What are the combined, long-term effects of climate change, air pollution and forest use on biodiversity in the forest landscape of southern Sweden?
- Evaluate the relative and individual long-term effects of:
 - climate change,
 - air pollution
 - forest use
 - geochemical soil characteristics

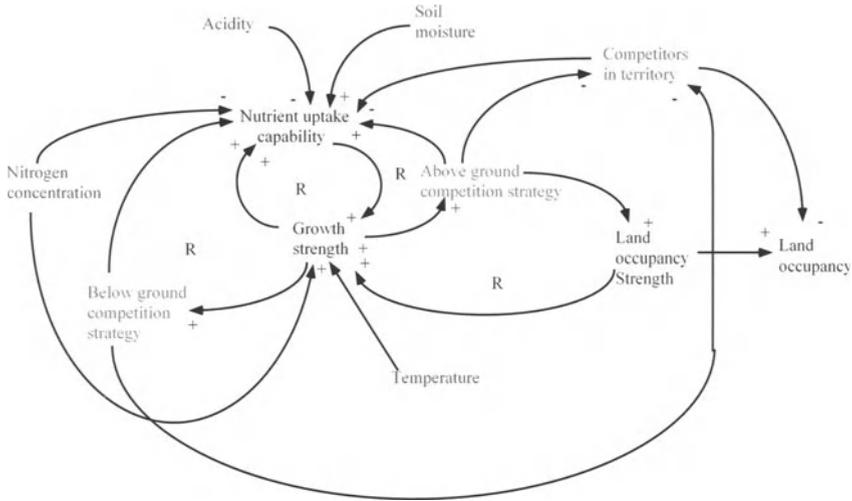


Figure 9.6: A causal loop diagram for ground vegetation dynamics with respect to ground cover and response to pollution and climate change.

– the integrated effect of climate change, air pollution and forest use on biodiversity in the forested landscape of Southern Sweden.

- Estimate the critical load of acidity and nitrogen to achieve a certain limitation of environmental damage.
- Identify especially sensitive geographical areas and ecosystems as well as how many areas of high biodiversity is needed for maintenance of the overall biodiversity of today?

Due to resource and time limitations in the SUFOR programme, specific components of the total biodiversity system were highlighted:

- Occurrence of wood-living insects species
- Occurrence of lichens species
- Forest tree species

For the biodiversity of wood living insects, the influence of dead wood, tree species composition, forest structure defined as tree age as well as occurrence of fires in the forest were identified as crucial factors. These components were also analyzed in relation to landscape structure. The system is illustrated in the causal loop diagram of Fig. 9.8. This figure also show the causal loop diagram of the total system of insect biodiversity.

9.4.1 Ground vegetation modelling

Several approaches have been tried for modelling of vegetation changes caused by pollution and management impacts (van Dobben et al. 2001; Schouwenberg et al. 2001; Wamelink et al. 2001). Such changes can be reinterpreted to implications concerning changes in biodiversity of the ground vegetation. Such models have been focused on the species with higher ground

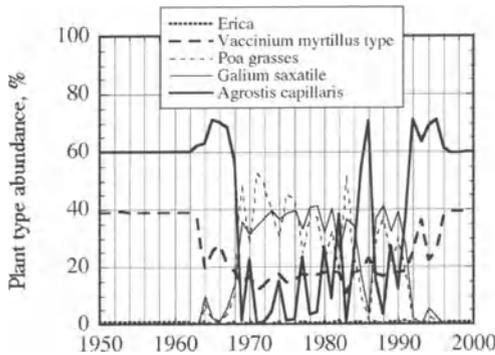


Figure 9.7: Preliminary calculation for the Fårahall research site at Hallandsåsen, southern Sweden. The changes in ground vegetation at this forest stand with Norway spruce and Scots pine is mainly driven by acid deposition effects, nitrogen inputs and forest stand stem density which all changed during the simulation period.

after 1990. The response is initiated by the onset of increased nitrogen deposition and the second change by recovery from soil acidification and nitrogen pollution. The changes in ground vegetation at this forest stand with Norway spruce and Scots pine is mainly driven by acid deposition effects, nitrogen inputs and forest stand stem density in this run, they all changed during the simulation period. The change in the nitrogen input which first increased from 5 to 25 kg N ha⁻¹yr⁻¹ and then declined to 12 kg N ha⁻¹yr⁻¹. Acid inputs increased from 5 kg S ha⁻¹yr⁻¹ to 35 kg S ha⁻¹yr⁻¹ and then declined simultaneously with nitrogen to 15 kg S ha⁻¹yr⁻¹. In this particular case, competition was included for nutrients, but not fully for competition for light. Recovery of non-present plant species was set to occur instantaneously once conditions allow for it. No change was assumed for climate during the period. The plant species used should be seen as functional classes which have for the sake of experiment borrowed their properties from the species mentioned. The changes in ground vegetation at this forest stand with Norway spruce and Scots pine is mainly driven by acid deposition effects, nitrogen inputs and forest stand stem density in the model (Hansson 1993). In this particular case, plant competition was included for nutrients, but not fully for competition for light. Recovery of a non-present plant species was set to occur instantaneously once conditions allow for it.

9.4.2 Biodiversity of insects

In SUFOR, much emphasis was placed on species diversity of wood-living insects which constitute only a part of the whole insect biodiversity system (Fig. 9.8) in the landscape, and many simplifications have been made.

9.5 Conclusions

Assessments of change and modelling of biodiversity needs to take into account the importance of the structure and feedback systems in the ecosystem. This can only be done by system analysis

coverage or by assembling plants into groups with predefined properties and functions. A ground vegetation model is being developed (Hansson 1995), the basic outline of its basic principles have been outlined in Fig. 9.6. An example of what a ground vegetation model may yield is shown in Fig. 9.7, where results from applying the preliminary VEG model has been shown (Hansson 1995). Similar models are being developed at the ALTERRA institute in Wageningen, Netherlands (The SUMO-SMART2-NTM system). The scenario shown is a preliminary calculation for the Fårahall research site at Hallandsåsen, southern Sweden. An early model version with response functions for acidity, nitrogen and competition was used. The diagram shows the response of the mainly changes in soil pH and nitrogen inputs assumed to have started in earnest around 1960. The deposition was assumed to decline

mate the whole ecosystem biodiversity. Without models, the presently available information cannot be properly sorted, nor used for future predictions.

We must keep clearly in our minds the present perspective of a very dynamic future, as well as of a very dynamic near history. In the next 200 years, the world population will increase by 50%, some say more, the world will adopt energy production systems not based on oil, the climate will change significantly, as well as the pollution situation will change. Acidification of soils by deposition of S and N will be strongly decreased, land eutrophication by N deposition will be somewhat reduced, but it will continue and the future climate will be significantly warmer and wetter than now in most parts of Sweden. We will get many aspects of the climate at presently existing in the northwestern European continental sea border (Great Britain, Northern France). All this makes predictions of future biodiversity very difficult and impossible to do without good models. Most is uncertain, but we can be certain of this; the biodiversity of our landscapes will be in a constant state of change for at least the next 300 years, most likely forever, it will certainly not become what it was once, historically, neither it will remain as it is today, and it will evolve into something significantly different. The problem of biodiversity change is in this perspective not a phenomenon that can be made to stop, for biodiversity to be conserved, but rather a problem of managing a process of change towards desirable endpoints. The inescapable conclusion is that we will need both goals for future biodiversity, and estimates of possible new endpoints under free landscape development, but more importantly, for different scenarios of landscape management and utilization.



Figure 9.10: *After a forest fire, pioneering tree species such as birch, cherry, aspen and rowan initially colonize the ground. These stands are succeeded by larger deciduous hardwoods or by large conifers. Photo: M. Svensson.*

9.6 Principles of biodiversity preservation in forestry

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9.6.1 Introduction

Biodiversity is the variety or diversity of life (Hunter 1999). Here we define biodiversity as the genetic variation within species, species richness and variation of forest types/forest ecosystems with an emphasis on species richness. Genetic variation within species and species richness of microorganisms is outside the scope of this discussion. The single most important factor causing a loss of biodiversity is human land-use (Nowicki et al. 1998). Sustainable forestry implies maintaining biodiversity at a high level, and where it is eroded or destroyed restoration is an essential component of the sustainable use of forests (Frelich and Puettmann 1999). This applies in particular to Western Europe, where very little of natural forests remain (Hannah et al. 1995, Peterken 1996). The basic principles of biodiversity preservation in sustainable forestry build to a large extent on general principles of biodiversity preservation. From several recent analyses and reviews (e.g. Gaston 1994, 1996, Lawton and May 1995, Kunin and Gaston 1997, Hanski and Gilpin 1997, Hanski 1998, Hansson 1997, McCarthy and Lindenmayer 1999, Hanski and Ovaskainen 2000)) we can extract some general principles (Nilsson and Niklasson 2001):

- The probability that a species is present increases with the amount of suitable habitat.
- The extinction risk of a local population increases strongly with diminishing population size.
- Species with large territories need a minimum amount of suitable habitat at the landscape level, either all in one large patch or grouped in several small patches situated near enough to permit movement between the patches.
- Several nearby patches of a habitat increase the probability that species dependent on that habitat are present. Thus, isolated patches have a lower probability of being occupied.
- In a group of habitat patches, the long-term survival of a local population can depend on patches that at a given time are unoccupied.
- Extinction of a local population can occur many years after the amount of suitable habitat in a landscape has decreased below some critical threshold.
- As a consequence of the previous principle, the most cost-efficient restoration of habitat is to increase the habitat(s) that has decreased most recently and to increase habitat near biodiversity hot-spots.
- Species that occur on many sites in a landscape are often common at the sites where they are present. Thus, rare species are often locally rare apart from occurring at few sites.
- Species occurrences are often "nested", which means that at sites with rare species the common ones are also present.
- The risk of local extinction increases with population variability, habitat specialisation and low dispersal ability.
- Many species are dependent, directly or indirectly on natural disturbances, such as fire, wind-throws, water fluctuations and grazing by large herbivores.

- Disturbances due to man can replace natural disturbances only if they resemble natural disturbances in critical aspects (e.g. by creating key structures like large snags, downed dead wood and burnt dead and living trees of many species). The disturbance process in itself does not guarantee a preserved biodiversity.

These basic principles should be integrated with what is known about the natural biodiversity and disturbance regimes of a region, to arrive at some principles of biodiversity preservation. Therefore, after discussing the historical aspects of biodiversity we give a short review of what is known about natural disturbance regimes in southern Sweden (Nilsson and Niklasson 2001).

9.6.2 Definition of sustainable biodiversity

Sustainable biodiversity in Sweden is defined in the SUFOR programme as the maintenance of viable populations of all indigenous organisms occurring in the country. This definition must be interpreted, since the spatial scale of biodiversity preservation is not stated. We interpret it as the preservation of viable populations of species in all regions where climate and forest types allow their existence. A regional approach is central for efficient biodiversity preservation. Since the climate is changing we must now and then revise our species lists, and one aspect is the possibility of bringing back species previously exterminated by forestry. The concept of minimum viable populations is evolving which also means a constantly updated monitoring of population sizes etc. We still know very little about this aspect for a majority of species (e.g. Nilsson et al. 2001). Clearly much remains to be done in terms of monitoring and research before we know if present forestry practice is compatible with a sustainable biodiversity.

9.6.3 Establishing the baseline of forest biodiversity

Sustainable biodiversity is in Sweden defined as the maintenance of viable populations of all indigenous species in their former distribution ranges. However, establishing which species are indigenous and what are their natural distribution ranges constitutes a problem. Is the current distribution of species in equilibrium with climate and vegetation zones, or is it largely determined by the recent fragmentation and degradation of their habitat? These questions can only be answered by obtaining hard data on the past distribution of plants and animals and their response to various degrees of human impact upon their habitats, from moderate cultural modification to degradation, fragmentation and outright destruction. In order to be able to assess and counter the current threats against biodiversity we need a long-time perspective, where anthropogenic influence can be distinguished from natural causes such as climate changes or natural vegetation successions (Nilsson 1997a, Cowling et al 2001). This may pose a problem, since the modern analogues of many past habitats are more or less drastically altered or even nonexistent. With today's growing awareness that the increasing human impact causes an accelerating loss of biodiversity, a new field of applications for palaeoecological methods is emerging. The most important palaeoecological method for reconstructing vegetation history is pollen analysis (Prentice 1988). In several recent studies, "floristic diversity", i.e. the number of pollen taxa, has been used as a measure of biodiversity (e.g. Birks and Line 1992, Odgaard 1994, 1999, Lagerås 1996). However, "pollen diversity" will mostly reflect the *landscape diversity* around the sampling site: a landscape including hay meadows, fields, forest clearings, hedgerows etc. will automatically yield a higher number of pollen taxa than a more homogenous forest landscape. In this way, small-scale human impact will, by increasing the habitat diversity, almost inevitably result in an increase in floristic diversity, regardless of what the effect has been upon the diversity of the different habitats themselves. Furthermore, studies of recent conservation forests has shown that floristic diversity, although it may indicate the degree of openness and former grazing pressure in a forest, is not correlated with the species richness of tree-dependent lichens and beetles (Nilsson et al. 1995). In addition, factors of vital importance for biodiversity such as the age structure of the forest and the presence of dead, dying or hollow trees and coarse woody debris on the ground remain inaccessible to the palynologist. Depending

on the type of management structures such as old, dying trees, hollow trees or dead wood may be totally obliterated, even if the forest remains standing and no evidence of change can be seen in a pollen diagram. On the other hand, many species can find refuge in ancient trees in the cultural landscape, in environments that bear little palynological resemblance to ancient forest (see below). These reasons have made insect analysis (palaeoentomology) an increasingly attractive instrument in palaeoenvironmental reconstructions (Brayshay and Dinnin 1999). The great value of insects as palaeoenvironmental indicators lies in the fact that a large number of taxa can be identified to species level, and that many species have very narrow habitat tolerances. This allows the analyst to make detailed inferences about the past environment from the habitat preferences of the species present in the sample. Furthermore, insect analysis offers direct knowledge of a group that is both numerically and ecologically a very important part of the diversity of forest ecosystems.

9.6.4 Structure and biodiversity of the natural forests of Europe

Since the Early Neolithic, European woodlands have been affected by deforestation, girdling, coppicing, pollarding, shifting cultivation, grazing and other human activities (Pott 1992). In Western Europe today, virgin forests in a strict sense are nowhere to be found. Even forests *resembling* a pristine state are limited to a small number of isolated "habitat islands", almost always in places where topography, hydrology or low productivity has made agriculture and forestry impractical or uneconomical. The representativity of these small patches can often be questioned, due to their small size and isolation, and their atypical geology or topography. An important issue in conservation biology is thus how the structure of the woodlands has changed as a result of human impact, and what impact this has had on biodiversity. The most striking difference between natural forest and managed forest is the age structure of the tree population. Natural forest is far more dominated by old and very old trees than managed forest, and there is an abundance of old, dying trees, hollow trees or dead wood of various tree species and in various degrees of decomposition. The amount of dead wood can be at least ten times that of managed forest (Warren and Key 1991, Nilsson et al. 2001). The pre-cultural European forests are often depicted as continuous, dark and impregnable, but there are several reasons to think that this picture is incorrect. Notwithstanding the fact that shores of lakes and rivers, wetlands, cliffs and areas with very thin or infertile soils have always provided open spaces, gaps will also have been created by the death of old trees, insect outbreaks or catastrophic events such as storms or forest fires. Although forest regeneration is usually rapid, it seems likely that more permanent openings have been maintained by large herbivores such as deer, bison and before their extinction also auroch, wild horse and forest elephant (Owen-Smith 1987, Nilsson and Ericson 1997, Bengtsson et al. 2000, Vera 2000). For example, recently burnt forest provide an abundance of nutritious saplings, forbs and grasses that should have been highly attractive to these large herbivores. Beaver colonies in their movement along streams produce open flooded meadows surrounded by numerous dead trees, and it has even been suggested that early man may have used these for his initial colonization of the forests. Rackham (1998) gives many examples of both man-made and natural semi-open tree-lands in Europe (corresponding to the savannas of other continents), and stresses their importance for biodiversity preservation. To summarize, the processes mentioned above created a mosaic of different successional stages in close juxtaposition in the virgin forest. Probably as a result of this, many forest species have evolved to depend on both ends of the successional spectrum. Insects whose larvae depend on dead wood frequently need open space as adults for mating and dispersal, and abundant flowers producing nectar and pollen as a source of energy and protein (Warren and Key 1991). How the various evolutionary histories of different deciduous tree species may have influenced the habitat preferences of their respective beetle fauna is discussed by Gärdenfors and Baranowski (1992). Many red-listed saproxylic beetles and epiphytic lichens are today mainly found in/on old solitary trees or groves in the wooded pastures or hedgerows of the cultural landscape (Harding and Rose 1986, Nilsson et al. 1994, Rackham 1998). One might question whether these species are indigenous forest species.

We believe that the original disturbance regimes of the past, with frequent forest fires, flooding near large rivers and lakes and/or grazing maintained a dynamic, partly open forest landscape. For the species which have evolved in these open forests, the cultural landscape is one of the few environments that offers suitable habitats today.

9.6.5 Human impact in the forests of Europe

Most palaeontomological data on the biodiversity of pre-cultural forests comes from British studies. The presence of a virgin forest element in Early- to Mid-Holocene beetle faunas was first demonstrated by Osborne (1965), and has later been confirmed by others (e.g. Buckland 1979). The persistence of the virgin forests has been a matter of debate. Earlier beliefs that fragments of primary forests may have survived as late as the seventeenth century has been replaced by the realisation that human impact upon the woodland was considerable already in prehistory (Buckland and Dinnin 1993). However, records from several sites suggest that, in spite of the extensive forest clearance commencing in the Neolithic and continuing throughout the Bronze Age and Iron Age, relatively undisturbed woodland long persisted in agriculturally marginal areas (Dinnin 1997, Whitehouse 1997). Finally, natural forest disappeared almost completely from Britain (as from most of Europe), and it is hardly surprising that the forest fauna has suffered greatly (Buckland and Dinnin 1993). Osborne (1997) lists a number of wood-living beetles from the Neolithic, including such species as *Rhysodes sulcatus*, now extinct over most of Europe. Buckland and Dinnin (1993) list 29 species of beetles recorded from Holocene deposits in Britain, but now extinct from the area. No less than 17 of these are associated with timber and/or dead wood. Other elements of the forest fauna survive only in scattered fragments or in poor analogues of their original habitats (Speight 1989). All in all, among British beetles the evidence for regional extinctions due to climatic change during the last 1000 years is rather unconvincing, while the evidence for extinction due to anthropogenic influence is overwhelming (Wagner 1997).

9.6.6 Differences between southern Sweden and northern Europe

Berglund et al. (1991a), Nilsson (1997a, 1997c) and Lagerås (1997) describes the history of the cultural landscape in southern Sweden. In this area, the deforestation (with the exception of southern Scania) has been neither as early nor as complete as in Britain, but nevertheless the cultural impact in the forest has been considerable for 6000 years (Berglund et al. 1991b). In southern Scania, the virgin forest was transformed to a more open forest containing shade-intolerant trees around 5100 BP, while a major deforestation occurred around 2700 BP (Regnéll 1989). There are considerable regional differences in the timing and intensity of human impact even between various parts of southern Sweden. Human impact was delayed in inland/upland areas compared with more fertile lowlands (Berglund et al. 1991b). In the uplands of central Småland (Lagerås 1996), land use was long limited to wood pasturage, until a major deforestation transformed the landscape into a semi-open pastoral landscape (ca 0-500 AD). Around 500 AD an agrarian regression resulted in reforestation of abandoned pastures, a situation that persisted until a second agrarian expansion and deforestation took place in the Middle Ages (1000-1400 AD). This resulted in a diverse cultural landscape with pastures, hay meadows, permanent arable fields and some slash-and-burn cultivation. Several studies suggest that in some areas considerable human impact may have been even later than indicated, and that the species-rich deciduous forests may have persisted locally until only a few hundred years ago (Björkman 1997, Björkman and Karlsson 1999, Lindbladh 1998). During the last 150 years, slash-and-burn cultivation and hay production has ceased, grazing has decreased and coniferous forest has expanded at the expense of open land (Nilsson 1997a, 1997c).

9.6.7 Palaeontomological studies of biodiversity in Sweden

In Sweden, recent palaeontomological studies have largely been focused on the climatic development and faunal successions of the last glacial cycle (e.g. Lemdahl 1988, Gaillard and Lemdahl 1994, Coope and Lemdahl 1995, Lemdahl 1997); but there are also several studies of the insect fauna in archaeological contexts (e.g. Lemdahl and Thelaus 1989, Lemdahl et al. 1995). Although recent palaeontomological studies contain several records of saproxylic beetles, even extinct species such as *Rhysodes sulcatus* (Lemdahl, pers. comm.), comprehensive studies of the Fennoscandian beetle fauna of middle to late Holocene forest environments are scarce. Recently, beetle diversity and forest history during the last 1,000 years have been studied on several sites in southern Småland (Ljungberg, unpubl.). In spite of the short time frame compared with most other palaeontomological studies, also these records contain species now locally or even regionally extinct. In one of the sites, 23 species of wood-living beetles today included on the Swedish Red Data List (Gårdenfors 2000) have been recorded in peat deposits from the last 1000 years. Of these, no fewer than 16 are today extinct from the area. Considering the relatively high biodiversity of the site today, this number is surprisingly high. The extinct species are mainly those associated with old deciduous trees and coarse woody debris in late stages of decomposition. In accordance with the statements above also the early deciduous forest (where little or no evidence of human impact can be discerned) contained species depending upon sun-exposed habitats.

9.6.8 Disturbance regimes of southern Sweden

Due to the long history of human impact on southern Swedish forests it is extremely difficult to draw conclusions on what is the "normal" disturbance regime based on the present situation. Therefore, it is necessary to analyse past conditions using different techniques for reconstruction: pollen analysis, dendrochronology, macrofossil analysis and historical accounts (see preceding section, as well as Eliasson and Nilsson 1999, 2001, Hannon 1999, Lindbladh 1998, Niklasson 1998). From the present we can however draw some important conclusions based on records of abiotic factors like storm frequencies and incidence of natural fires (e.g. Lightning Ignition Density (Granström 1993)). Wind is a very important disturbance agent that cause both large- and small scale disturbances. It is very difficult to draw conclusion of what is "natural wind-fall frequency" based on the present patterns of wind felling. This is due to the fact that the structure and stability of natural forest is completely different from the production forest. The same reasoning accounts for fungi and insects that may have mass outbreaks in some years. Because of the massive changes in tree species composition and forest structure that has taken place due to forestry and changes in land use the last 100-300 years (Nilsson 1997a, 1997c), the present population density of so called pest species may be higher than long ago. Examples could be some insects and fungi, e.g. the bark beetle *Tomiscus piniperda* and the fungi *Heterobasidion annosum*. The disturbance regime of a landscape is the result of a complex mix of factors. Lightning Ignition Density (Granström 1993) and the relation and allocation of landscape elements like lakes, bogs, watercourses (Niklasson and Granström 2000, Hellberg 1999) is probably largely decisive for the resulting fire frequency, and for southern Swedish conditions especially: human impact (Niklasson and Drakenberg in press). As already mentioned, storm and hurricane frequencies (Henry and Swan 1974, Falinski 1986, Pontaville et al. 1997) are important factors both at a large and a small scale by felling and breaking trees. Less studied but probably as important for the disturbance regime is the soil moisture and texture by giving different stability to trees and stands to wind (Falinski 1986) and by providing conditions for different field vegetation and fuel for fire. Often neglected when discussing disturbance regimes are the tree species themselves, which may display very different properties in resistance to rot, regeneration strategies, fire resistance and stability to wind, life span etc. Such factors have large influence on the disturbance regimes and the population dynamics that may be important to consider when planning for sustainable forestry. Impact from grazing and browsing animals

on successions (Bergqvist 1998, Kullberg 2000) have been largely neglected as well as the effects of grouting by wild boar (Welanders 2000). The ecological effects of the beaver were probably also prominent. In the following we will give a short overview of the palaeoecological data available for reconstruction of the natural disturbance regime of the nemoral and boreonemoral (Sjörs 1963) zones of southern Sweden. The largest disturbance agent at present is forestry in south Swedish forests, approximately 3-5% of the forest area is annually affected by different forestry activities like thinning and clear-cutting. Although there are some similarities to wind disturbance, there are some fundamental and indisputable differences, above all the removal of biomass (mainly wood) from the forest ecosystem. Another largely neglected factor is that extremely few trees are allowed to fulfil their natural life cycle because rotation times are considerably shorter (typically 80 to 120 years) than the maximum life span of most tree species (from 150 to more than 800 years). The fundamental consequences of this for biodiversity are described below.

Fire

Fire is probably the single disturbance agent with the largest impact on the dynamics and composition of past south Swedish forests outside the nemoral zone. Compared to wind disturbance, the past situation is much better known for fire due to the direct evidences found in biological archives (charcoal, fire scars in trees and stumps). The impact on vegetation and the importance of fire for biodiversity is described in Heliövaara and Väisänen (1984), Skog and Forskning (1991), Ahnlund and Lindhe (1992) and Wikars (1992, 1997). A more extensive review of fire regimes is given by Granström et al (1995), and human use of fire is summarised by Larsson (1995). Today, the annual area burnt by forest fires amount to a few hundred hectares in southern Sweden (Anon. 1981). This equals to about 0.01% of the area, which of course is very little compared to the area disturbed by forestry annually. However, the few exactly dated south Swedish fire histories so far published or under work show that 2-5% of the forest area burned annually in the past (Niklasson and Drakenberg 2001, Niklasson and Karlsson 1997, Page et al. 1997, Niklasson et al. 2001). Today, most fires start because of human activities (neglectance, power lines, trains, camp fires etc) but Granström (1993) showed that ignition by lightning can contribute up to 25% of the fires and that the highest lightning ignition densities were found in Southeast Sweden. He also showed a strong gradient over south Sweden in fire ignition density, with low figures in the west and high in the east, very much following the general trend in humidity. In the humid province of Halland in southwestern Sweden, fires ignited by lightning are rare ($0.0-0.0003 \text{ km}^{-2} \text{ yr}^{-1}$), while in the dryer climate of early summer in the provinces Kalmar and Östergötland in southeastern Sweden fires ignited by lightning occur 7 times more frequent ($0.0021-0.0024 \text{ km}^{-2} \text{ yr}^{-1}$; Granström 1993). Palaeoecological studies of fire use mainly tree rings or charcoal in peat and lake sediments. The two methods largely supplement each other. Tree ring records are remarkably accurate for dating fires, but covers rather short time frames (-600 years in S. Sweden). Charcoal studies in sediment are rather inaccurate but span over very long times, usually over several millennia. In southern Sweden a further complicating factor, compared to northern Sweden, is that forestry and other activities normally have actively removed old trees and wood with fire-scars. Therefore, most studies with tree-ring reconstructed fire histories are restricted to areas with low productivity and human population density. Fire histories with tree-ring material of *Pinus sylvestris* from southern Sweden display very frequent fires compared to northern Sweden. In Tiveden National Park (Page et al 1997, Niklasson et al. in press), the mean fire interval was about 20 years from early 1400s to the start of fire suppression in 1850. In Kvill National Park, Northeast Småland (Niklasson and Drakenberg 2001), the results were similar with very short fire intervals, but with a much earlier suppression of the fires, beginning already around 1770. In Murstensdalen, eastern Värmland, fire intervals were somewhat longer but still short: 30-40 years with fire suppression since early 1800s (Niklasson and Karlsson 1997). In Hornsö kronopark in eastern Småland, fire intervals were around 30 years with the last large fire in 1868 (Niklasson and Gustafsson 1999).

In Siggaboda, south Småland, frequent fires were recorded 1555 to 1752 and no fires after that. From the Asa case study area, frequent fires were recorded at one point in the park during the 17th century. It is worth mentioning that in 1652, four of the six studied areas in southern Sweden had experienced large fires. This fire year has also been recorded frequently in northern Sweden and must have been a truly exceptional summer in terms of large forest fires. It is clear that the conifer-dominated studied areas in South-Southeast Sweden all have experienced fires 2-5 times more often than reported from north Sweden (Kohh 1975, Niklasson and Granström 2000, Zackrisson 1977). Fire intervals of 20 years are probably near the intrinsic shortest time required for the vegetation to recover and be flammable again (Schimmel and Granström 1997). This implies a strong human impact on the fire regime. Slash-and-burn agriculture and burning for improving *Calluna* heathland is well documented in many parts of southern Sweden but burning in forested areas is a much less known practise of the past land-use regime (Weimarck 1953, Larsson 1989, Granström 1993). To differentiate between natural and human fires is of course very difficult, but many indirect factors point towards a widespread use of fire in forest by humans, probably for improving grazing. A large discrepancy in number of fires per area unit compared to natural levels (Granström 1993) is evident in Kvill and Siggaboda, but it is difficult to draw conclusions since the studied areas are generally too small to permit such comparisons. In Tiveden on the other hand there was no visible change in fire regime when humans entered the area in the 17th century, suggesting a large natural component in fires. During the earlier times individual fires were very large, which indirectly support that naturally fires prevailed (Niklasson and Granström 2000). With these results as a background, it is clear that a controlled re-introduction of fire for biodiversity restoration purposes into South and Southeast Swedish forest ecosystems is strongly supported by the past situation and is urgently needed from a conservation point of view (see below). The highly improved conditions for sexual regeneration of most post-fire deciduous trees, *Betula*, *Populus*, *Salix*, would also significantly support in restoring the tree species composition towards more natural conditions.

Deciduous forest

In the nemoral zone and in the deciduous-dominated forests in the boreonemoral, our knowledge of the past situation is even more restricted and confined largely to charcoal records in sediments. In general, charcoal (=fire) in sediments is much more rare in deciduous forests (Karlsson 1996, Hannon 1999) in the southwest part of Sweden than in southeastern forests (Lindbladh and Bradshaw 1998). This trend is in accordance with the levels of natural lightning ignition (Granström 1993), but it also reflects the general difficulty or almost impossibility of igniting fuels in a deciduous forest. However, charcoal is not completely lacking even in the moistest rich deciduous forests and strikingly often charcoal is found in connection to large shifts in the tree species composition (Karlsson 1996, Hannon 1999, Björkman and Bradshaw 1996, Iversen 1941). The only reasonable explanation for this is human use of fire for clearance or crop-cultivation etc. A practical experiment conducted in the 50's in Denmark fully supported the possibility to burn deciduous forest during certain conditions (Steensberg 1993, Iversen 1958). Still, wind in combination with fungi should be the major factors driving the dynamics and governing the composition of deciduous forests in the south. However, the frequency and intensity of windfall events are unstudied in this part of Scandinavia. Such studies would be valuable, especially in connection with biodiversity assessment studies, for improving guidelines for sustainable management in deciduous forests. In natural deciduous forests the main disturbance factor apparently should have been wind, creating the typical small-scale gap phases (e.g. Emborg et al. 2000). The effect of different mix of deciduous and coniferous trees on wind disturbance patterns should be a priority for research in southern Sweden. The oak, *Quercus robur*, is probably a tree that has a high fire adaptation, but this is so far only deduced from American studies and observations (Abrams 1992), and the notion of high *Quercus* pollen levels in south-east Sweden in the past (Björse et al. 1998). The general regeneration failure of oak in dense forests is well known (Löf 1999, Vera 2000), and also documented in south Swedish forests

(Nilsson 1985). Frequent fires of a low intensity certainly should have maintained forests sparse enough for the light demanding oak-regeneration. Recently, it has been proposed that the oak is strongly favoured in areas with continuous and heavy grazing (Vera 2000). This may very well explain the occurrence and dominance of oak in southern Sweden in some areas, but not in the whole region. On the contrary, heavy grazing in combination with slash and burn agriculture seem to have severely decimated the oaks in most Swedish grazing commons ("utmark") 200-300 years ago (Eliasson and Nilsson 1999, 2001). The reason may be that slash and burn agriculture also eliminated thorny shrubs (cf. Vera 2000). To conclude, it is rather clear that fires have been present and frequent in many parts of south Sweden, mainly in the eastern parts but also in the north-central part. The high frequency of fires is difficult to imagine, but definitely many forests were more open than at present due to fires. The role of fire in most deciduous-dominated forests has probably been very marginal; instead wind, water and grazers must have been the major disturbance agents. Oak is probably an exception to this, since much indirect evidence suggest it has a rather efficient adaptation to fire. Deeper studies into the disturbance regimes of southern Sweden are called for, both in deciduous, mixed and coniferous forests. References of past forest composition, disturbance frequencies and intensity of disturbance in all forest types, preferably combined with parallel biodiversity assessments and experimental fire studies, is highly needed for further development of sustainable forest management models.

9.6.9 Critical features for biodiversity preservation in forests

This section is based on the review by Nilsson et al. (2001a). As pointed out above, fire has had an important role in the natural disturbance regimes in southern Sweden. However, although it is important to understand the connections between landscape features and fire frequencies, the relation is rather loose (Granström 2001). Thus, in our view, the ASIO model of forest management in boreal forests (Angelstam 1997) will solve few of the most urgent problems with biodiversity preservation. These are more connected to key structures created by disturbances instead of to the disturbance process per se. Therefore, we stress the importance of sufficient amounts of burnt trees, old trees (>150 years) and large dead trees (diameter >40 cm) with different exposure in the landscape that are created in large amounts in natural landscapes by disturbances. Even in so-called "diversity-oriented silviculture" all the largest trees are removed (Lähde et al. 1999), with obvious negative consequences for demanding species.

Densities of large living trees

Large and old living trees are important for many species, especially among insects, fungi and lichens (Nilsson et al. 2001a). Reference densities of such trees ought to be established for different types of forests in different regions. In boreonemoral forests, trees with DBH above 70 cm may be considered as large trees, as also suggested for tropical forests (Clark and Clark 1996). Some measurements in boreonemoral forests in southern Sweden, eastern Poland and Slovakia indicate densities around 15 such large trees per ha in old-growth forests with a mixture of deciduous and coniferous trees (Nilsson et al. 2001b). In the same forests, densities of trees with larger hollows were 10-30 per ha, and no spruces and very few deciduous trees below a diameter of 40 cm contained hollows (Nilsson et al. 2001b). In managed forests in southern Sweden less than 1% of these reference densities of large trees occur at present (SNV 1999), Fig. 9.11. Note that the present densities of such large trees are higher in pastures than in forests managed for timber production. This probably applies to most of Europe, although we only have data from Sweden. For a restoration of biodiversity and establishing of viable populations of species dependent on old trees the densities of large trees must probably be increased to about 20% of reference densities (Nilsson et al. 2001a). However, very few relevant studies are available and more tests are obviously needed to get more secure results since this is one of the most economically demanding measures for forestry to accomplish.

Large dead trees

In general, forestry reduces the density of large dead trees much more than that of thinner trunks (e.g. Green and Peterken 1997, Kirby et al. 1998, Spetich et al. 1999). Cavity nesting birds often prefer standing dead trees with large diameters (e.g. Raphael and White 1984, Håggvar et al. 1990, Bunell et al. 1999). Further, many wood-beetles are dependent on standing dead trees and can not develop in downed logs (e.g. Palm 1959, Nilsson 1997b). Some of the more specialised wood-dependent beetles only live in large trunks that usually do not exist in managed forests. The habitat requirement of several species in nemoral and boreonemoral forests suggests that a diameter of 40 cm may be a critical lower limit for the most demanding species living in dead trees (Nilsson et al. 2001a). Thus, for maintaining biodiversity big trunks are crucial, because with the presence of large dead trees also smaller dimensions of dead wood will be automatically available. Furthermore, big trunks remain standing for longer than smaller trunks (e.g. Raphael and White 1984, Lindenmayer et al. 1997) and decomposition of their wood takes a longer time (Harmon et al. 1986, Stone et al. 1998). Even after long drought, there will remain some moist wood in the centre of big trunks preventing sensitive species from drying out. These various considerations have led us to the hypothesis that for a given volume of dead wood big trees can save more species from local extinction in managed forests than the same volume of thinner trees (Nilsson et al. 2001a). Some measurements in boreonemoral forests in southern Sweden, eastern Poland and Slovakia indicate densities around 30 dead trees with DBH above 40 cm per ha in old growth forests with a mixture of deciduous and coniferous trees (Nilsson et al. 2001b). About half of these large dead trees were standing, but only 14% among trunks with diameters between 20 and 40 cm. The present amount of large dead trees in south Sweden is less than 1% of the reference density (SNV 1999), Fig. 9.11. For a restoration of biodiversity and establishing of viable populations of species dependent on dead trees the densities of such trees must probably be increased to about 20% of reference densities (Nilsson et al. 2001a).

Burnt forest

At least 70 species, mainly insects and fungi, are directly dependent on burnt forest in northern Europe (Wikars and Ås 1999). Many more species in the boreonemoral forests are dependent on features and succession stages that normally follow after fire, e.g. deciduous successions dominated by birch and aspen (Nilsson and Ericson 1997). In the south-eastern part of Sweden, with a high natural fire frequency, a sparse pine- and oak-dominated forest with birches and aspen probably was the most common type of forest in the past (Björse et al. 1998, Lindbladh et al. 2000). In the area with the highest fire frequency even an open savannah type of forest may have been present. However, this is at present highly speculative since too few palaeoecological studies have been made that focus on these questions and that would permit us to draw general conclusions. It can also be suspected that wet and moist soil conditions could provide protection

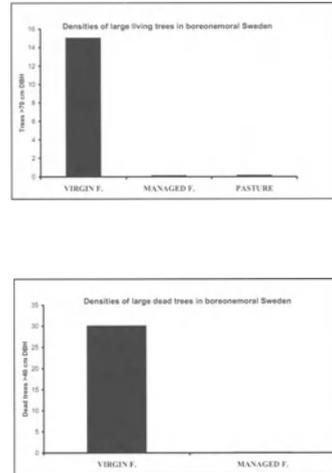


Figure 9.11: *Densities (No. per ha) of large living trees (DBH > 70 cm) in the boreonemoral part of Götaland, southern Sweden. Densities (No. per ha) of large dead trees (DBH > 40 cm) in boreonemoral Sweden. Note that the density is extremely low in managed forests but not zero.*

from fires where fire-sensitive deciduous species, e.g. *Fagus*, could survive. In other parts of the world, like in North America, oaks in temperate forests have been shown to be strongly promoted by fires (e.g. Abrams 1992). Unfortunately, reference forests with natural disturbance regimes are lacking, as are relevant scientific studies in Europe. Anyhow, fires were not only important as an agent providing many sun-exposed dead trees, but also by opening up the forest around old living trees. It is notable that most beetles living in old oaks are favoured by sun-exposure (Ranius and Jansson 2000), and the same seem to be true for red-listed pine-dependent beetles (Gärdenfors and Baranowski 1992).

Reserves contra managed forests

Biodiversity preservation can not only be based on features and substrates left in managed forests. The species most sensitive to transformation of the forest can only survive in large reserves in the long run (Økland 1996, Soulé and Sanjayan 1998). However, to prevent genetic impoverishment some exchange of individuals between such reserves may be needed. Therefore, some "stepping stones" with suitable habitat ought to occur in the managed forest too. For species adapted to burnt forest, relatively less amounts of burnt trees compared to the original situation may be needed to maintain the dependent species in the landscape. We base this suggestion on the high dispersal rate of fire dependent species, but due to the long and highly efficient fire suppression even some such species have gone extinct from Sweden. On the other hand, species dependent on features that are relatively stable over time like old living trees and large dead trees seem to have lower dispersal propensity (Nilsson and Baranowski 1997a, Ranius and Hedin 2001, Hedin 1999). For example, a beetle, *Ceruchus chrysomelinus*, living in large dead trees may be lacking in apparent suitable habitat only 1-2 km away from occupied patches (Nilsson et al. 2000). Recent population studies on the fauna connected to old, hollow trees suggest that the spatial distribution of the trees is important, and that they should occur in groups in order to be of the highest value for biodiversity preservation (Ranius 2000, Ranius and Wilander 2000). Groups of old living trees may also be better than widely scattered trees for preservation of cryptogams in managed forests (Hazell and Gustafsson 1999). Unfortunately, little information about the effect of different distribution patterns of dead trees is available. However, two species of beetles living in basidocarps of *Fomes fomentarius* have higher frequency in clumps of habitat (Sverdrup-Thygeson and Midtgaard 1998, Kehler and Bondrup-Nielsen 1999). This may be a general feature among many dead-wood insects (Rukke 2000, Schiegg 2000). These studies support the recommendation that groups of dead trees are better than the same number of trees spread out in the landscape. Thus, we conclude that retention of patches of old living trees and dead trees may be most cost-efficient if they are positioned in larger stands or groups. One possible exception is retention of natural edges between wetland and forest. To leave such ecotones may be useful even if they are long and narrow, because water disturbances in combination with dead wood is necessary for some species (Nilsson and Baranowski 1993). However, relevant studies of green tree retention strategies in nemoral and boreonemoral forests seem to be lacking regarding these ecotones.

Indicator species of a high biodiversity

Recently, a large-scale survey of forests in Sweden have used indicator species among vascular plants, cryptogams and fungi for finding habitats presumed to contain red-listed species (Nitare and Norén 1992, Hansson 2001). However, we know of no studies validating these species as indicators, with the exception of the lichen *Lobaria pulmonaria*. Forest stands where this species occurs have higher number of red-listed lichens and also more red-listed beetles living in hollow trees (Nilsson et al. 1995). However, the number of other red-listed wood-beetles was not correlated with presence of this lichen. The present designation of habitats valuable for biodiversity preservation in Sweden suffers from a lack of indicator species dependent on open forest habitats. Open forest habitats with older trees are today very rare in managed

forests because of their intrinsic counteracting impact on forest wood production. Rundlöf and Nilsson (1995) and Nilsson et al. (2001a) suggested some possible indicator species that are dependent on open forest habitats. In general, due to their generalised habitat requirements vertebrates may be less suitable than other organism groups as indicators of a high biodiversity in forests (Nilsson et al. 2001a). However, evidence is accumulating that large carnivores are important for a high biodiversity because they depress populations of smaller carnivores and herbivores (e.g. Soulé and Terborgh 1999). Without the large carnivores, predator-prey interactions are disrupted threatening many species. Therefore, the presence of large predators (wolf, lynx and bear) in the landscape may be one component of sustainable use of forests. However, they are hardly suitable as indicator species (Linnell et al. 2000). With the growing interest in sustainable forestry, reliable biodiversity assessment methods must be developed. A large discrepancy has occurred between the biodiversity preservation strategies applied and their evaluation. Thus, practise has run far ahead of validation tests of proposed indicator systems. This unfortunate state of affairs ought to be rectified. The costs for green tree retention are high and ecological and economical research should definitely make this action more cost-efficient. The general presumption that retaining and creating important structures for biodiversity in all landscapes, irrespective of their biodiversity status at the present time, will also result in a high biodiversity should be tested. If forest continuity and substrate history is important for retaining a high biodiversity in nemoral and boreonemoral forests, as suggested by Nilsson et al. (2001a), structures alone may not be appropriate indicators of a high biodiversity. Forest continuity means the presence over long time (two or more tree generations) of features e.g. shrub and/or tree cover (tree continuity), ancient or big trees (ancient tree continuity) or big dead trees (låga continuity) (Nilsson and Baranowski 1993, Nilsson et al. 1995). In the highly transformed forests of southern Sweden, species may be better indicators than structures, but this crucial suggestion must be tested and evaluated. Such tests are expensive, because sampling and species identification of diverse groups of forest living organisms are necessary and the only way to obtain reliable answers to these questions. If appropriate resources are not allocated to validation tests we will continue with methods of uncertain reliability. This may prove very expensive in the long run causing unnecessary species extinctions.

Deciduous trees and biodiversity preservation

The proportion of spruce in the forests of southern Sweden has increased over a long period of time, and the present high proportion is one of the most important reasons for the present biodiversity crisis (Nilsson 1997a). For example, the Lesser Spotted Woodpecker *Dendrocopos minor* requires at least about 20% older deciduous trees within its large territory (Wiktander et al. 2001). An increased share of deciduous trees in the forests in this region may therefore seem to be an obvious solution. However, such a development can only solve a few of the problems. Species such as many vascular plants can benefit from more light reaching the forest floor in deciduous and mixed forests compared to spruce forests. However, if deciduous forests replace spruce forests the colonisation rate of many herbs may be too slow to recolonize the new deciduous forests for hundreds of years (Brunet and von Oheimb 1998). Therefore, the continuity of the present deciduous forests and the planning for and establishment of new deciduous forest near colonisation sources may be equally, or even more, important factors for biodiversity preservation than just focusing on the proportion of deciduous forests in the landscape. The acidification of the upper soil is partly due to the high proportion of spruce (Bergkvist and Folkesson 1995, Nilsson and Tyler 1995). This part of the soil and its surface layer is a very important habitat for snails, which are sensitive to low pH and calcium content. In recent decades, snails seem to have totally disappeared from coniferous forests on oligotrophic soils (Gårdenfors et al. 1995). This reduction of biodiversity may have important consequences for other organism groups. In the Netherlands, it has convincingly been shown that a lack of snails in the forests can impair the reproductive success of birds (Graveland and Drent 1997). The laying birds need snails as a calcium supplement when breeding. It is very likely that the same problem occurs in

southern Sweden. If an increased proportion of deciduous trees in the forests can increase pH and calcium content of the soil surface snails may recolonise former coniferous forest. However, the problem with a low colonisation rate of snails is the same as for herbs (see above). The most promising aspect of an increased proportion of deciduous trees in the forests is that green tree retention may be more cost-efficient when the forest is cut. Since old and dead deciduous trees are more important for biodiversity restoration and preservation in southern Sweden compared to coniferous trees (Nilsson and Niklasson 2001), increased opportunities of leaving deciduous trees at final cutting occur with more such trees in the managed forests. As pointed out above, the present extremely low density of old deciduous trees and large dead trees are perhaps the most severe problems for biodiversity restoration and preservation in southern Sweden.

Natural regeneration and biodiversity preservation

One of the major hypotheses of the SUFOR programme is that an increased area with natural regeneration is one of the solutions for a sustainable forestry. In terms of biodiversity preservation there are few reasons to object. However, there are some possible pitfalls. If coniferous forests are regenerated below a rather dense stand of seed trees (shelter-wood), spruce may increase its dominance in future stands more than if the same forests were regenerated by clear felling. Thus, in such cases natural regeneration may even be negative for biodiversity preservation in the long term. On the other hand, in deciduous and mixed deciduous/coniferous forests natural regeneration may be an efficient way to increased biodiversity in the future. However, this depends very much on how the regeneration measures are performed. For example, if spruces are left as seed trees the regeneration can be dominated by that species. Conversely, if all spruces are cut and deciduous trees are retained the stand can develop into an important site for biodiversity in the near future. Such measures are crucial for restoration of biodiversity over large areas of southern Sweden, where only one or a few percent of the forest area at present supports a high biodiversity. Probably, this proportion must be increased 5-10 times for preservation of viable populations of all forest living species, due to long time lags in local extinction. Such extinctions are expected because huge areas have been transformed from mixed forests with deciduous trees to spruce dominated as recent as in the last 50 years. Furthermore, the forests have become much denser in the last century with negative consequences for many species (e.g. Berg et al. 1994, Nilsson and Baranowski 1997b, Ranius and Nilsson 1997).

9.6.10 Recommendations for biodiversity restoration

When managing habitats with trees, many measures can be changed to improve biodiversity preservation and restoration. We have recently given recommendations for habitats with trees in southern Sweden (Nilsson and Niklasson 2001). These recommendations are translated here

- Known centres of high biodiversity connected to trees, listed in Nilsson and Niklasson (2001), should be preserved. Suitable habitats for the most demanding species in each centre ought to be restored in the near surroundings.
- Inventories in order to identify additional centres of high biodiversity should be started, mainly using well-known indicator groups such as epiphytic lichens and wood beetles. Such inventories are also important to design effective management plans for each area.
- Biodiversity considerations in forestry should be adapted to the more demanding species known to be present in the landscape.
- Tree species composition in conservation forests should be managed, so that species that have decreased most in recent centuries are increased. In southern Sweden this means deciduous trees, especially oaks *Quercus spp.*, lime *Tilia cordata*, alder *Alnus glutinosa* and hazel *Corylus avellana*. When cuttings are performed, this gives an opportunity of increasing the amount of coarse woody debris.

- EU-rules for wooded pastures must be changed to encourage the retention of old trees and their successors and also leaving dead trees and heaps with branches in sun exposed situations.
- Coniferous trees and young deciduous trees that threaten the survival of old trees (>150 years old) should be cut or pollarded. Relevant tree species are mainly oaks, lime, elm, maple and ash. In many areas with species dependent on old trees there is a lack of trees of these species in the 100-200 year age interval. The continuity of the crucial habitat is dependent on survival of the ancient trees in the coming century. Only branches below a diameter of 10 cm (occasionally 20 cm) should be cut on old neglected pollards. Using this recommendation, the crown can be reduced on old pollards to prevent wind breaking of the tree. A few branches must however always be left. Cutting of all branches that is often done at present in Sweden will kill many of the old pollards.
- The amount of coarse woody debris should be increased at all management measures. When large trees must be cut, e.g. for safety reasons, high stumps ought to be left together with the trunks in suitable places. Such sites should preferably be sun exposed , e.g. along power lines.
- Flowering shrubs, e.g. *Crataegus spp.*, *Salix caprea*, *Prunus spinosa*, *Rosa spp.* and *Viburnum opulus*, should be retained and planted where absent both in pastures, protected and managed forests.
- Natural fire disturbance should be introduced in several very large forest reserves (several 1000 ha), especially in southeastern Sweden. Accidentally burnt forests ought to be left uncut. A strategy for reintroducing fire in forests ought to be developed.
- The "key habitat" inventory should be complemented with habitats that today not are considered as "forest land" in the Swedish law. Such habitats, especially wooded pastures, harbour the majority of the populations of threatened insect and lichen species dependent on old trees in Sweden.
- Ditched wetlands in forest habitat should be restored as wetlands to increase the amount of dead trees and landscape heterogeneity.

9.6.11 Conclusions for a sustainable forestry

In terms of sustainable forestry some measures are especially important for biodiversity restoration and preservation:

- Restoration of more natural disturbance regimes in many forests is crucial in developing a sustainable forestry. In southern Sweden this means increasing the use of fire in forest management, including forest reserves. To be a really useful measure only stands with many, or preferably all trees left should be burnt.
- The densities of very old trees (>150 years) must be increased strongly, e.g. probably more than ten times the present density in southern Sweden.
- The densities of large dead trees (DBH>40 cm) must be increased very strongly, and in southern Sweden to more than ten times the present density.
- Naturally occurring forest types and trees that have decreased most in recent centuries should be highest priority for restoration.
- Restoration measures should preferably be positioned within dispersal distances of demanding organisms (1 or a few km) of known local "hot spots" with high biodiversity.

Chapter 10

Forests, Acidification and the Socio-economic Cost

Christer Kalén, Bengt Nihlgård and Harald Sverdrup

10.1 Introduction

The continuing anthropogenic acidification of forests soils in Sweden is a serious threat to the longevity of these utilities. Anthropogenic activities can thus lower the total value the forest represents, which in turn would have a negative impact on the welfare of Swedish society. The acidification of soil results in the leaching of plant nutrients (Falkengren-Grerup 1995) such as potassium, calcium, and magnesium, which in the long term can cause nutrient deficiencies (Thelin et al. 1998), threatening the productivity of forest soils. The process of acidification also results in elevated concentrations of dissolved aluminium and other toxic metals in the soil, ground water, and surface water (Berggren et al 1990; Johansson et al 1995). Biodiversity in lakes and rivers in areas affected by water acidification is seriously impoverished (Brodin and Henriksson 1995). Acidified ground water can cause problems of various sorts, such as the by corroding of pipe-work (Bertills et al. 1989) and the creation of health risks (Bjertness and Alexander 1997), since the acidification increases the mobility of various harmful metals such as aluminum, mercury, copper, zinc, cadmium, and lead (Johansson 1995). Acid deposition accelerates the deterioration of building materials as well as of objects of art and of historical interest, particularly in urban areas (Kucera et al. 1996). The understanding of how forests are affected by acid deposition has improved significantly and it has become clear that many values in society are threatened by the ongoing acidification of Swedish forest soils (Johansson et al. 1999). There are measures of several kinds, apart from the lowering of the acid deposition, which can be carried out to mitigate such effects. Liming and fertilising with wood ash are two such measures that have been in focus within recent decades. Although these measures can prevent many of the negative effects of acidification, they cannot restore the soil to its natural condition. Liming and fertilising mitigates the impact of acidification and thus improves the conditions of the soil. Since mitigating forest soil acidification involves certain costs, it is relevant to explore the extent of these costs to assess if they are socio-economic justified before action is initiated on a national scale. To judge whether such countermeasures are economically justified, one first needs to estimate their total economic costs. From an economic perspective, the costs of mitigation measures should be less than the total economic losses that would otherwise result. Many factors make studies of this sort complex and difficult to carry out, however. For example, it is very difficult to estimate the long-term effects on the different utilities involved. Converting the effects on e.g. the environment or on cultural heritage, to economic costs is also very complicated. The economic impact of a decline in forest production depends on such factors as when decline would occur and its magnitude. If we knew the volume of timber lost, it would be

possible to estimate future impact on costs through use of today's timber prices. Other utilities, such as those of a recreational character, as well as biodiversity and ecological services, are more difficult to set a current economic value upon. Estimating future negative impacts there can be there can be trickier. One reason for that is that many of the effects involved have no current market value. There are two types of uncertainties that need to be multiplied in estimating the future economic costs of acidification, namely the ecological and the economic.

$$\text{Total Uncertainty} = \text{Ecological Uncertainty} \cdot \text{Economic Uncertainty} \quad (10.1)$$

The uncertainties are complicated further when the political and the social domain are included. The ecological and the economic impact of different actions are partly dependent upon the political success of enforcing restrictions so as to alter the social behaviour that caused the problems in the first place.

The aim is to explore the possibilities and drawbacks of using economic valuation to establish the basis for deciding what measures should be carried out for mitigating a future negative environmental effects. In this connection, various values that are negatively affected by forest soil acidification are examined. The expected costs of not undertaking measures to mitigate forest soil acidification needs to be compared with the costs and economic gains of measures to mitigate these effects. The cost of liming and the effects of acidification are considered in this paper. Other effects, such as eutrophication, are outside the scope of this paper, although acidification may indeed lead to an increase in nitrate leaching from forest soils. In addition, both the abatement costs of reducing the production of acidifying pollutants at source and political activities aimed at combating acidification, such as the signing and ratification of protocols under the Convention of Long Range Transboundary Pollution (CLRTAP), are omitted.

10.2 Ecology and Economics

The interaction between ecology and economics has become an important focus of study recently. The interest in studying causality relations between economics and the ecology of forest has grown since it has been noted that economic activities often result in negative environmental effects and that this has serious implications for sustainability. Any economy depends on the environment as a source of life-support services and of raw materials. Unfortunately, neither the market nor a planned economy takes account of the entire value of the goods and services involved, or of the costs borne by society if the supply of natural resources is reduced or the services are impaired, either now or in the future (Folke et al. 1993). Harold Hotelling (1895-1973) developed a model of the efficient use of resources over time, one that helps to explain how natural resources are driven to degradation or even extinction (Norgaard 1995). According to Hotelling's model, even when market prices fully reflect the value of a natural resource, it is economically efficient under some circumstances to degrade an ecosystem or exploit a species to extinction. Hotelling's logic was quite simple. If the value of a natural resource does not increase as rapidly as the rate of interest, both individual owners of the resource and society at large are economically better off in exploiting the resource more quickly and putting the returns the exploitation has brought in the bank. These returns can then be invested in the creation of humanly produced capital that earns a return greater than the rate of interest. In this economic view, natural resources are a form of natural capital that can be converted into human-produced capital and should be so converted if they fail to earn as much as the human-produced capital does. This reasoning not only describes why economically rational owners of natural resources exploit them to extinction, but also prescribes that the owners "should" do this. Thus, as long as we assume that markets reflect true values, both historical and ongoing losses of genetic, species and of ecosystem diversity are efficient and "should" occur. Hotelling's reasoning currently dominates resource economic theory and the policy advice from economists (Norgaard and Howarth 1991). Environmental economics has been developed since the late 1960s and

today we have a better understanding of interactions between the economy and the environment. New disciplines have also been founded by persons who argue that the present economic discipline does not take adequate account of environmental degradation (Turner et al. 1994). Ecological economics is a trans-disciplinary approach to environmental sciences that examines the interdependent relationships between ecological and economic systems as well as between mounting global environmental, population, and economic problems. The overall objective is to sustain both ecological and economic systems by identifying ways by which local and short-term goals and incentives, such as those of local economic growth and advancement of private interests, can be made consistent with global and long-term goals (such as sustainability and global welfare) (Costanza, 1996). The basic world view of conventional economics is one in which individual human consumers are the central figures. Their tastes and preferences are taken as given and are the dominant determining force. The resource base is viewed as essentially limitless, due to technical progress and infinite substitutability. In ecological economics, scientists from different areas meet side-by-side to develop methods, tools and models that address different aspects of sustainability (Costanza et al. 1994). They adopt a holistic and systemic view and they believe it to be necessary to go beyond the traditional disciplinary borders. Neo-classical economics distinguishes three different types of capital: natural, human and manufactured capital (Costanza et al. 1994). Ecological economists regard these different types of capital as being complementary to each other, which is different from conventional national economics, where these are regarded as exchangeable (substitutable). Natural capital is regarded within ecological economics as a limiting factor for further development. It is believed within ecological economics that a minimal demand for sustainability is to maintain the natural capital stock at its present level. It is therefore better to improve the effectiveness with which we extract and refine the natural resources rather than increasing the actual withdrawal. For forestry, this implies that the volume production (growth rate) should not be regarded as the best option to increase the economic value. A more sustainable strategy would be to improve the efficiency with which we handle and refine timber.

10.2.1 Accounting the Environment

The gross national product (GNP) of a country as measured in terms of its national accounts is commonly used as an indicator of a nation's wealth. If the GNP increases, the society's welfare is assumed to increase. Quantifying the national accounts of is a complex task and no attempt will be made to clarify details of the theory and practice regarding it. Certain aspects on the matter, however, are of interest here. How, for example, should non-marketed goods and services be dealt with? Hultkrantz (1991) asks in this connection, how one should handle a natural reserve. The establishment of a natural reserve is designated in the national accounts in Sweden today as a cost. The exclusion of productive land from commercial use results in a loss of economic profits which otherwise would have been included in the national accounts. On the other hand, the reason for establishing a natural reserve is that of society's regarding the forest as being more valuable than its being used for timber production. Establishment of the forest reserve thus increases the national wealth by providing for more appropriate use of the area

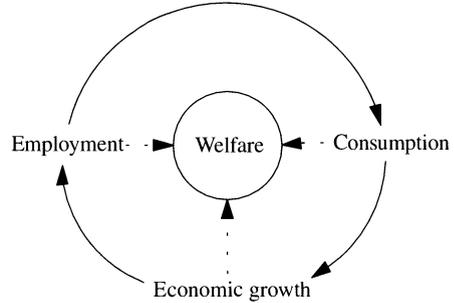


Figure 10.1: *For most people, the general perception of a society's development and welfare seems to be regarded as a self-reinforcing loop. By stimulating consumption, economic development increases. An increase in economic development increases employment.*

in question. In the present national accounting system, it is nevertheless identified as a cost. GNP is a measure of economic activity and one way of increasing the activity is to stimulate the consumption within the society. Thus, an increase in economic activity has direct effects on the GNP and lead to perception of an increase in wealth (Fig. 10.1). When economic development occurs, unemployment usually decreases, which is one of the most important political goals. More people having a job also stimulate consumption. These three components reinforce each other and drive economic progress forward. With an increase in economic development, a general environmental degradation usually occurs due to an increase in the emission of pollutants and in loss of natural habitats. Pollution or degradation of the environment reduces societal welfare without necessarily affecting economic activity (Fig. 10.2). One reason for this is that many values are not incorporated into the GNP, despite their contributing to welfare. Assigning economic value to natural resources is very important since one often values societal welfare in economic terms. If an economic activity transforms an overall loss of value into a net environmental asset but the losses as such are neglected, this can give a false picture of the increase in wealth. The problem of valuing the loss of natural resources or damage to them has gain considerable attention internationally, many countries endeavouring to improve their national accounts by inclusion of these factors. By signing the Agenda 21 document in Rio de Janeiro 1992, Sweden committed itself to developing their environmental accounts in a manner taking better account of the connections between economic activities and environmental degradation. If one can adequately conceive the impact on the environment of some action or state affairs in monetary terms through converting environmental degradation into economic costs, this allows the information about such degradation to be integrated into the national accounting system. This can help improve the manner in which environmental effects are dealt with in political decisions.

It would seem appropriate to consider some critical views towards the practise of "setting a price on our environment". First, it forces us to set an economic value on assets and utilities, even if they have no present value on the conventional market. For example, if one converts the value of biological diversity into an amount of money, it can be argued that this is neither practical nor ethically justified. Also, it may be very difficult to deal with the future in this way. In the present economic system, we set an economic value on, or simply produce future environmental effects - effects that not we but future generations have to pay. These future generations have no opportunity of intervening in the process. Thus, we put ourselves above future generations and there is no appropriate economic (or democratic) method to prevent this from happening, except that of moral commitment.

In addition, the establishment of environmental accounting can lead to one's relying, and becoming dependent upon measuring invaluable goods and services in economic terms. It is difficult to set an economic value on many of the utilities connected with the environment. It can also be questioned whether it is appropriate to do so in light of moral and ethical standards. An example of this would be to place an economic value on some other species, on the risk of an environmental catastrophe or on people's manner of experiencing natural scenery or the beauty of nature. There are several arguments that could be put forward, nevertheless, to justify the development of an environmental accounting system. Politicians base many of their decisions on economic information, and environmental degradation might be disregarded if such costs were omitted. If policy makers

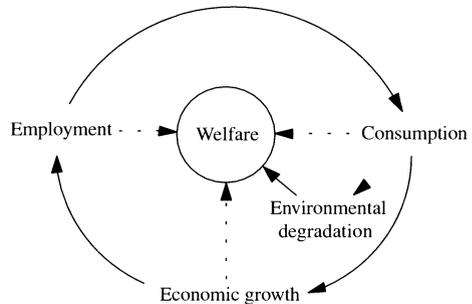


Figure 10.2: *Economic development is often connected with an increase in environmental degradation that in time results in costs to the society and thus negatively affects societal welfare.*

need to make trade-offs, it is essential to know what is being traded-off against what. Unless one has some idea of the economic value of an environmental asset, the basis for making an appropriate decision is insufficient. Environmental accounts, despite their drawbacks, provide one picture of how one utilises one's natural capital and how it is degraded. For example, if one hectare of forest is felled and sold on the market, this natural capital is converted to capital in monetary terms. One can ask whether anything has been gained in this conversion. Natural capital has decreased and monetary capital has increased. Since forests are a renewable resource and, if managed properly, the same conversion can be performed again after some time. Sustainability implies that this conversion can be carried out forever. For oil or iron ore the situation is different since there natural capital which is not renewable is being extracted. It can thus be argued that in principle the conversion of non-renewable resources leads neither to an increase nor a decrease in the net national product. The net national product (NNP) is thus a more accurate measure of the value of a nation's productions since it includes the total change in capital stock. The task of evaluating in a fully appropriate way total value of a forest would require a massive workload and represent a job that could never be completed.

10.2.2 Socio-economic justification

From a socio-economic standpoint, a justified measure can be described as an investment that increases the total national welfare if it is carried out. The liming of lakes can serve as an illustrative example. Roughly 100 million SEK is spent annually on the liming of lakes and streams (Ahnér and Brann 1996). For this activity to be considered socio-economically justified, the total economic value of receiving water of improved quality needs to be as high or higher than the cost. A problem here is how one should estimate the economic value of the quality of water being improved. Lakes have value in various respect that are more or less connected to each other. Not only does the commercial value of the fishing industry improve with a measure of this type, but also the value of recreational fishing improves, which is of considerable value both for Swedes and for tourists from abroad. It is generally easier to estimate the positive effects on the fishing industry using conventional economic methods than to evaluate the total value derived from recreational fishing. One should point out that approximately 50% of the total harvest is attained through recreational fishing (SCB 1995). It is difficult as well to estimate the economic value of an increase in biological diversity or how future generations will appraise an improvement in water quality. Four different types of costs in an environmental accounting system can be distinguished: damage costs, restoration costs, mitigation costs and avoidance costs (SCB 1997). Each of these is described briefly below.

1. **Damage costs** designate the economic consequences of avoiding countermeasures to mitigate the impact of an environmental problem. For our purposes, these costs refer to the economic consequences of acidification of the forest soil continuing.
2. **Restoration costs** refer to the estimated costs of restoring a damaged area to its original status. It is usually difficult or impossible, however, to restore a damaged ecosystem to its original status.
3. **Mitigation cost** concern situations in which it is impossible to restore an area to its original status. Such costs, which can be defined as those of restoring the environment to an acceptable level, depend on the level one considers acceptable.
4. **Avoidance costs** represent the costs of preventing future damages from arising. These costs include those of minimising the risk of a future impact on an ecosystem by such means as a change in management, for example. In our case, these costs relate to the different alternatives that are available for mitigating the risk of future damage costs by improving the capacity of the forest soils to prevent acidification.

Mitigation costs are referred to hereafter in discussing alternative measures that can be undertaken to restore already affected soils to an acceptable level and thus reduce the risk of a future impact.

10.2.3 Impact of the Discount Rate

The discount rate further complicates the estimation of future damage costs. Even if one concluded that the liming of forest soils would be economically worthwhile, it is not certain that this holds true when the discount rate is included. Cost estimates are dependent on the mitigation costs, the damage costs, the time prior to impact and the discount rate. A measure of particular value carried out today assumes a higher value in the future. Consider a bank investment of one SEK with 3 percent interest rate. In ten years, the value will have increased to 1.34 SEK, and in 30 years it would be 2.43 SEK. This is because of capital produced in the bank at the present interest rates. A measure that cost one million SEK today has to prevent a costs of 2.43 million SEK after 30 years, to be considered worthwhile. In our example of preserving forest productivity, the cost of losing some future amount should be discounted to the present value of it before it is compared with the present cost of mitigation. A forest used commercially has a life cycle of approximately 70-120 years, making the discount rate an extremely important factor to consider. The reason for using the discounted value is the belief that all forms of capital - natural, human and manufactured - are substitutable. One could therefore compensate a future generation's loss of natural capital by capital that has been produced. As mentioned earlier, ecological economists do not share this view and argue that the natural capital stock is fundamental and represents that on which all other economic activities rest. According to their view, natural capital is only complementary to human- and manufactured capital. It is not possible therefore, to unconditionally compensate the loss of natural capital by capital of other types. They argue therefore, that natural capital has to at least be kept intact at the present level. According to this line of reasoning, the discount rate should not be considered in attempting to justify a mitigation cost.

10.2.4 Estimating the Economic Value

An economic value is supposed to be a measure of people's wants and desires. In economics, the words "benefit" and "cost", respectively are used to indicate whether if a value is positive or negative. When people feel better off, a positive economic value arises, and when they feel worse off a negative economic value arises (e.g. Perman et al. 1996). Thus, an economic value only measures human preferences. Furthermore, it can only measure present (or past) preferences. Future preference, that is, how future generations allocate their values, cannot be measured today. In other words, an economic value is an anthropocentric tool for measuring the preferences of present generations. Placing an economic value on a natural resource thus only describes how we allocate our economic means and excludes by definition both other species and future generations of human beings. Measuring the economic value of natural resources can thus be hazardous since many stakeholders are excluded. Preferences are derived from wants and desires, and perhaps also needs, which are dynamic and are dependent upon the individual and the present situation. Economists often explain these fluctuations as being due to interactions between supply and demand. When the supply of a utility is in excess, the demand is low. The value that accrues for each unit of the utility has a lower value than it would have if the supply were lower than the demand. Thus, a natural resource is generally valued more when it is scarce. Values, or preferences, are also dependent to a certain extent upon knowledge. If we knew today that a specific utility would be scarce in the near future, the value would usually rise. Thus, knowledge of the future affects present value. Economic valuation of a natural resource can thus be criticised since it does not include appropriate consideration of the possible lack of knowledge. A natural resource can be said to represent a certain total value. When measuring the economic value of a natural resource from a society's point of view we should measure the

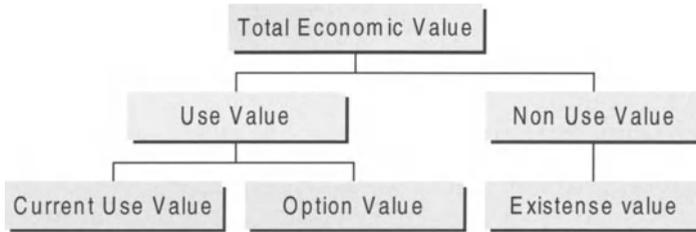


Figure 10.3: *The total economic value of a natural resource involves values of different types.*

total economic value (TEV). Different stakeholders, for example, benefit from forests in different ways. A resource thus has many values and as many of these as possible should be considered when estimating the TEV, which can be compartmentalised into different types of values (Fig. 10.3).

TEV can be divided into two broad categories: use value and non-use value. The use value can be further divided in current use (for forests, for example, forestry, recreation, carbon fixing, etc.) and optional use value (e.g. establishing a natural reserve). The non-use value of something is usually defined as the *existence value*, reflecting people's allocation of value to the knowledge that a specific resource exists, even if they never will use it themselves. The existence value of biodiversity is often referred to as a non-use value.

10.2.5 Methods for Valuing Natural Resources

Different methods have been developed to measure the economic value of non-marketed goods and services. The most frequently used are the travel cost method (TCM), the hedonic pricing method (HPM) and the contingent valuation method (CVM) (Wibe 1994). The travel cost method uses the revealed preferences for a utility, as represented by the capital spent to actually benefit from the resource involved. For example, the value of a national park can be estimated by use of this method through the total amount of money used when visiting the park being determined. Since people live at different distances from the park, they have different travel costs. By observing the number of trips made during a given period and the differing travel costs involved, a demand function can be identified. The Hedonic pricing method explores factors that affect the market value of a good. The most common application of this method is to the private property market. The market price of an estate is dependent on factors such as location, size, number of rooms, neighbourhood, etc. This method can be used to examine the extent to which different utilities contribute to the total price. For example, if a house is located close to a recreational area, it may be higher in value than if it were located close to a waste dump or a highway. By determining the value that can be credited to the presence of the recreational area, one can get an idea of its value. The contingent valuation method (CVM) is an approach that is popular today. This method involves conducting interviews to find out what people are willing to pay for a utility, for example a walk in the forest. An average value is then calculated to reflecting the value people attached to walking in the forest. By using hypothetical questions, the method can also be used to estimate the value of intangibles, such as the existence value of biodiversity. A problem with measuring the value of non-marketed goods and services, apart from methodological problems, is that it is questionable whether a value of this sort should be included in the national accounts. The main reason for this concerns the problem of allocation. Since non-marketed goods and services do not compete on an open market, they are not compared with utilities that have a particular market value. On the other hand, it can be argued that goods on the market have a form of shadow price since they are not

competing with the non-marketed goods and services. It should be noted that it is inappropriate to add the value of different CVM studies and to thus obtain a total value. Studies of this sort involve the assumption that all other factors are equal (*Ceteris paribus*) (Bostedt 1995).

10.3 Environmental Accounting in Sweden

Work at establishing an environmental accounting system started in Sweden in 1990, when the government decided to set up a committee on the matter. In 1991, the results were presented in the study "Räkna med miljön" (SOU 1991). The government gave two important directives to the committee. The first was to judge whether it was possible and appropriate to establish national accounts concerned specifically with natural resources and environmental issues. The directive stemmed from the UN's recommendation to include complementary environmental accounts in current national accounts. The second directive was to specifically study the meaning of sustainable development. Since 1987, when the Bruntland commission coined the term, ensuring future generations that they would inherit a level of national wealth no less than that today became an important political issue. It was acknowledged that the commonly used measure of wealth - gross national product - was insufficient for determining whether future generations would have the same opportunities to fulfil their need as present generations do. The aim in establishing environmental accounts was to register changes in the environment in both physical and economic terms. Due to the importance of healthy forests for Sweden's welfare, increasing emphasis has been placed on establishing environmental accounts. Three governmental institutions are engaged in the process of establish such accounts: the National Institute of Economic Research (Konjunkturinstitutet, KI), Statistics Sweden (SCB) and the Swedish Environmental Protection Agency (SEPA). Values not directly visible on the market (of the forest industry) have been included so as to provide a better picture of the total economic value. A household's consumption of game meat, berries and mushrooms had not previously been converted into monetary terms and was thus not represented in the GNP. These values are now being studied more closely so that they can be included as monetary values in environmental accounts. Other examples are the ecological services forests provide in terms of carbon fixation, water purification and habitat for biological diversity, which are also converted into monetary terms (KI 1998). In the governmental committee studying the possibilities of establishing environmental accounts, efforts have been made to set an economic value on Swedish forests. Lars Hultkrantz assisted in preparing an appendix (Guld och gröna skogar) in which he discuss ways to include forest values not previously included in the environmental accounts (Table 10.1). He estimates that the total value derived from forests in 1987 amounted to 22,000 million SEK. This was 4 thousand million more than the amount included in the national accounts. The gross value of felled timber was used to estimate the value of the timber. The value of berries, mushrooms and game meat was calculated as being 1,520 million SEK or 7% of the total value of the forest. The losses of biodiversity and of nutrient stock (due to acidification) were included as costs of 600 million each. Carbon fixation contributed with a value of 3,800 million SEK. The value of non-marketed goods and services contributes with 19% of the total value. It should be noted that recreation value was not included in the study.

The annual forest production has a net value today of approximately 30,000 million SEK (Skogsstyrelsen 1998). The increment in biomass volume during 1997 was 92 million m³. The mean price for timber that was sold on the market was 326 SEK ⁻³. The mean gross value of the annual felling between 1992 and 1996 amounted to 19,500 million SEK (Skogsstyrelsen 1998). In 1997, the taxation value of standing forest and forest land in Sweden was 192,000 million SEK. The products derived from forests contribute by 14% to the annual value of exports in Sweden (SCB 1998). SCB estimates that 30 million litres of berries and 14 million litres of mushrooms were collected for household purposes during 1995. Bil- and cowberries (Blueberries and lingonberries' or more formally; *Vaccinium myrtillus* and *Vaccinium vitis-idaea*) constitute the major part of the berries that were collected. Game hunting and recreational fishing are

| Service/"non service" | Reinvestment | Stock |
|--|---|---|
| Timber, biofuel, etc. | Forest management | Timber stock |
| Harvest of berries | | Berry plants |
| Harvest of mushrooms | | Mycelia |
| Game hunting, game meat | Game management, costs in forestry, agriculture and traffic | Game population |
| Existence value on forest dependent plants and animals | Flora and fauna management | Conditions for species survival |
| Impact on hydrological cycles etc. | Measures that affects runoff | Timber stock, clear cuts, ditches |
| Carbon fixation | Forest management | Carbon storage |
| Buffer for acid deposition, nutrients | Liming, fertilising | Soil buffering capacity and content of minerals |
| Nitrogen discharge | Building of nutrient sinks | Nitrogen fixation capacity |
| Feeds for reindeers | | Lichen availability |
| Recreation | | |

Table 10.1: *Environmental accounts for forests in Sweden. Source: Hultkrantz, 1991*

considered to represent important recreational values. They are also a source of income for many landholders. For estimating the recreational value of forests, a number of studies using CVM have been carried out. According to KI (1996), approximately 373 million visits are made to Swedish forests annually. The total value of is estimated to be 19,000 million SEK per annum (Jämttjärn 1996). However, great variation in the results makes this figure only a rough estimate. The best guess can be seen as being somewhere between 5,000 and 20,000 million SEK per annum. The value has been shown to be higher in forests with mixed tree species (Mattsson and Li 1994), implying that the recreational value increases if deciduous trees are integrated into existing monocultures of coniferous stands. Patches and differences in age distribution also increase the aesthetical value. Wibe (1994) conducted a survey on valuation studies on non-wood benefits in forestry. A considerable number of studies are examined and are categorised in terms of recreational, hunting and existence values. He concluded that for all the studies as a whole the mean value for one day's recreation in the forest (hiking, camping, etc.) is around 200 SEK. Studies of the value of game hunting in Sweden have been carried out. A hunter is prepared to pay between 4,600 to 4,900 SEK (1991 year prices) for acquiring hunting license for the current season (Monitor 1994). This is only a part of the total hunting costs actually paid by the hunter during a hunting season. Since there are approximately 500,000 hunters in Sweden, this suggests the total value to be more than 2,000 million SEK annually. Discounting for the future appears to be considerable among hunters, since they were only prepared to pay 3500 for the season thereafter. The value for one day of hunting has been estimated to be 320 SEK (Wibe 1994). The total value per hunting day for the 500,000 hunters in Sweden would thus amount to some 160 million SEK. Assuming 15 days to be spent on hunting a year, this would amount to an annual value of over 2,000 million SEK. In KI's survey, the value of hunting was calculated to contribute to the total recreational value by approximately 2,000 million SEK per annum. Several studies of the economic value of biodiversity in Swedish forests have been carried out (Wibe 1994). Johansson (1990) found the willingness to pay for preserving 300

unspecified species living in forests to be 85 SEK per annum. The WTP increased with the hypothetical number of species preserved. In another study, people were found to be willing to pay 450 SEK for saving the white-backed woodpecker (Fredman 1994). These two studies reflect the inconsistency of either people's preferences of valuing different species or of the method employed.

10.3.1 Estimating the value of utilities affected by acidification

Within the forest ecosystem, there are basically four values that could be affected by acidification: forest production, recreational value, biological diversity and ecological services. Their economic impact is not clearly visible, however, since we have not experienced or identified a reduction in the timber produced or quantified the value of a loss in biodiversity.

The total economic value of forests can be estimated to be approximately 55,000 million SEK per annum. This figure is derived summing the annual value of the timber produced (30,000 million SEK), the estimated recreation value (19,000 million SEK) and the value of berries, mushrooms, game meat and carbon sink, which in Hultkrantz's report was estimated to 19% of the total timber value. Note that these figures estimate the annual values derived from forests. The value of total timber stock in Sweden is considerably higher. To determine whether it is economically justified to mitigate the acidification of forest soils, an estimate of how these utilities are affected (and of the future economic impact this has)

should be made. In addition, many secondary effects of the acidification of forest soils are apparent outside the forest ecosystem. The limits of the system are thus important to consider. By expanding the limits of the system, other costs can be identified that could increase the total damage costs. The amount of money worth spending on mitigating a future impact would then increase. The annual WTP for mitigating forest acidification in Sweden is estimated to be 375 SEK per person (KI 1996). This implies a total WTP of 3,000 million SEK per annum. Forest soils leach acid water to adjacent watercourses. Eventually, the run-off water ends up in lakes, resulting in an increase in acidity. In 1985, a national survey was carried out in which it was estimated that 21,500 of Sweden's 85,000 lakes were damaged to a level at which many organisms were unable to survive (Monitor 1991). An impact of this sort on lakes has an effect on recreational fishing, which is an important leisure activity in Sweden. The economic value of recreational fishing in Sweden is estimated to approximately 10,000 million SEK annually (Ahnér and Brann 1996). To mitigate the problem of acidified lakes, the government granted 1,600 million SEK to a Swedish liming programme to be carried out during the period of 1976 and 1995 (Svenson et al. 1995). The acidity in forest soils will proceed downwards through the soil profile as long as acidification continues. Eventually, acidified water will reach the ground water. Bertills et al. (1989) estimated the annual cost for the increased corrosion of water pipes to be 200 million SEK. Acidified drinking water may also lead to negative health

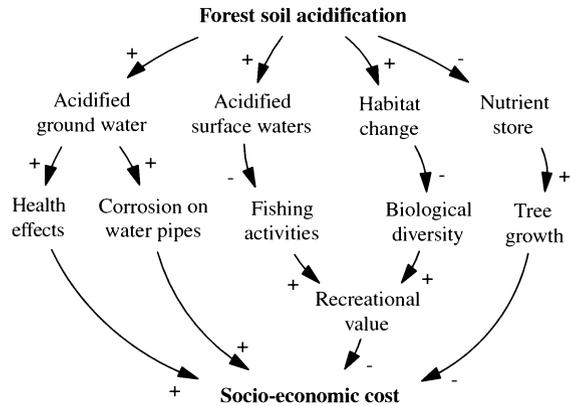


Figure 10.4: Forest soil acidification affects many values that contribute to the welfare of the society. Combatting the acidification of forest soils improves the conditions for values other than simply forest growth.

effects (Bjertness and Alexander 1997), although convincing evidence for this is lacking. The concentration of acid metal ions in water increases, especially aluminium, iron, cadmium and manganese, as pH values become lower (Johansson et al. 1999). In 1989, a total of 65,000 wells in southern Sweden were estimated to be below the recommended standard in terms of water quality (Bertills et al. 1989). Whether this is dangerous to health or not, people are worried about elevated concentrations of heavy metals in their drinking water. The total WTP of the adult population needed to maintain healthy drinking water has been estimated to 2,000 million SEK per annum (Silvander 1991). Acidification of forest soil will lead to secondary effects on the surface and ground water (including wells). Although there are presumably relatively few wells located in forests, it would seem relevant to include the effects of acidification on these values when estimating the costs of forest soil acidification. The liming of forest soils has been shown to enhance the conditions for both adjacent surface and groundwater (Nyström et al. 1995; Norrström and Jacks 1993). Restoring the pH in lakes will also improve the conditions for many species (Lingdell and Engblom 1995), although the recovery period may be considerable (Appelberg et al. 1993). We can conclude that forest soil acidification has a potential for affecting many values inherent to the forest ecosystem (Fig. 10.4). In addition, values that are found outside of forests are likewise affected by the acidification taking place within the forests. Thus, one can regard forest soils as a source of pollution that can affect the surface and groundwater negatively. The TEV of the utilities that have the possibility of being affected sums to 65,000 million SEK annually. This very considerable value should be born in mind when judging whether a mitigating measure is economically worthwhile.

10.4 Estimating the economically acceptable loss of value

Since a future cost due to forest soil acidification is difficult to estimate, one could study what percentage reduction in the annual value of stand growth is economically acceptable in order for liming to be socio-economic worthwhile. The annual value lost (c) is dependent on the estimated annual value (v) and the percentage of reduction (p) that is due to forest soil acidification, then;

$$c = \frac{v \cdot p}{100} \quad (10.2)$$

Assume that a specific measure mitigates a future reduction in growth for t number of years. The total value derived during this period can be written as:

$$TV = v \cdot t \quad (10.3)$$

The value of an investing in a mitigation measure is thus dependent on the time the mitigating effects will persist (i.e. how many years a decline in growth will be avoided).

$$p = \frac{100 \cdot c}{v \cdot t} \quad (10.4)$$

Future revenue, however, is discounted by a given rate (r), resulting in a total value that is lower than TV. If the discount rate is included, the equation becomes:

$$TV = \sum_t^1 v \cdot 1 + r^{-t} \quad (10.5)$$

With the same reasoning, the value lost (c) can be written as:

$$c = \frac{p}{100} \cdot \sum_t^1 v \cdot 1 + r^{-t} \quad (10.6)$$

Mitigating growth reduction is economically worthwhile up to the point at which the mitigation costs (m) equal the lost value that is discounted:

$$m = c = \frac{p}{100} \cdot \sum_t^1 v \cdot 1 + r^{-t} \tag{10.7}$$

Assume that liming would eliminate a future growth decline for t number of years. The cost of liming is 1500 SEK ha⁻¹ and under normal conditions the annual volume produced per hectare has a value of 2,200 SEK ha⁻¹. This is equal to:

$$p = \frac{100 \cdot m}{\sum_t^1 v \cdot 1 + r^{-t}} \tag{10.8}$$

where p indicates the percentage of growth decline required before a mitigation measure is economically worthwhile. This equation is similar to the first, except that the discount rate we is also included. A discount rate of 3 per cent is used in this example (r = 0.03). We here employ the estimated cost of liming, which is 1,500 SEK ha⁻¹. The annual value of the volume produced is estimated to be 2,200 SEK. In Fig. 10.5, a comparison of p is made between 1) a stand without use of the discount rate, 2) the total economic value (twice the stand value), and 3) a stand for which the discount rate is included. Since the cost of mitigating acidification by use of limestone (and wood ash) is fairly low (1,500 SEK per hectare), the

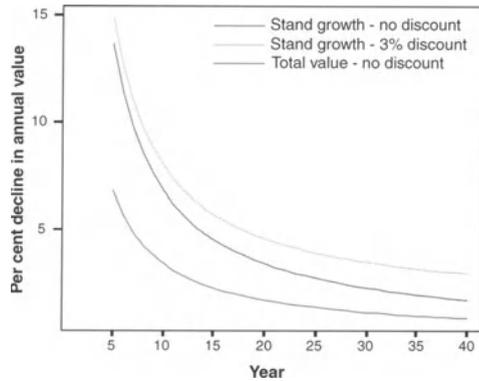


Figure 10.5: *The number of years a given mitigating measure prevails determines how much of a decline in value is justifiable to prevent a particular level of cost and a certain annual value.*

operation is worthwhile if the positive effects on tree growth last for a long period of time (20-40 years). A decline in forest growth of less than 5 per cent makes mitigation measures worthwhile, regardless of whether one uses a discount rate or not. A sensitivity analysis shows that the discount rate does not affect the pattern significantly. Furthermore, if the positive effects of liming will for a period of 20 to 40 years, the other two variables (costs of liming and value of the timber produced) will not have a major impact on the output. Thus, it seems reasonable to conclude that liming is economically worthwhile if it mitigates a future growth decline of 2-5 percent, and if the effect persists for more than 20 years.

10.5 Area Affected by Acidification

The economic cost of a future negative impact on forest production is dependent on the size of the area affected. An area of considerable size is, and will be, affected by acidification (Monitor 1991). According to Sverdrup and Warfvinge (1995), 80% of Sweden's forest soils exceed the critical load for acidity. When these areas reach a new steady state equilibrium, they will a lower pH and a lower concentration of base cations (BC). The critical load concept in these calculations

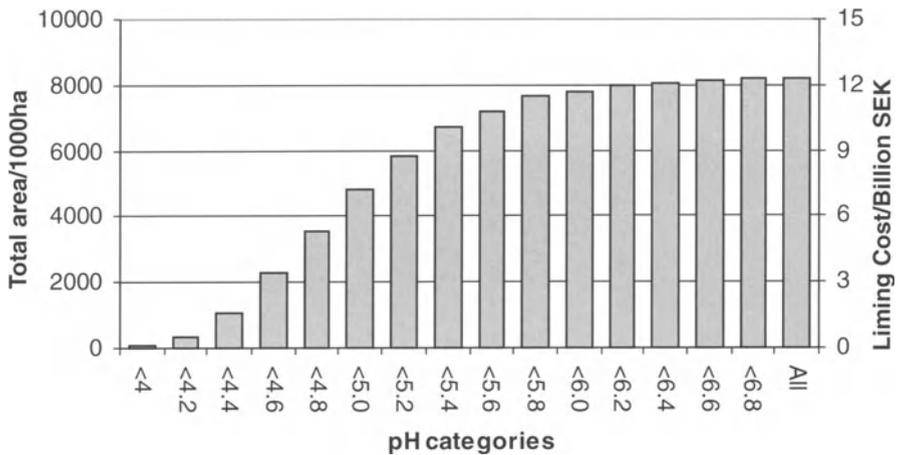


Figure 10.6: A chart showing the areas of forest land in southern Sweden below a particular pH value. Approximately 5 million hectares (or 60% of the total area there) have a pH value of below 5.0 in the upper B-horizon.

is defined as the maximum amount of sulphur and nitrogen deposition that will not cause long-term damage to ecosystem structure and function. The critical limit is the most unfavourable value the chemical criteria can attain without long-term harmful effects on the structure and functioning of the ecosystem (Barkman 1998). For forest soils, a $[BC]/[Al]$ ratio=1 is used as one critical limit (Sverdrup and Warfvinge 1993). If the ratio falls below this level, a reduction in the growth of trees can be expected. Data from the National Forest Inventory (NFI) were used to calculate how much forest land that lies within specific pH intervals. The present pH measurements in the B-horizon that NFI carried out were used for analysis. The reason for selecting this layer is that the upper layers are more variable and more dependent upon the age of the stand (Tamm and Hallbäck 1986). Although the B-horizon can be affected by stand cycle variations, the variations there are less pronounced than in the layers closer to the surface. In the NFI, the upper level of the B-horizon is sampled for pH analysis. The natural pH level (unaffected by human activities) there is believed to lie somewhere between pH 5.0 and pH 5.5 (Nihlgård et al. 1996). Calculating the mean cumulative value for each interval of the periods 1983-1987 and 1993 and 1995 that were sampled indicates approximately 60% (5 million hectares) of the total forested area in southern Sweden to have a pH value in the B-horizon of less than 5.0 (Fig. 10.6). Each sample in the NFI has a corresponding weight factor. This weight factor should be interpreted as an area around the sample in question that has similar conditions, allowing the sample to be represented by a particular area. The samples are assumed to correspond to the total forested relative area, which amounts to 8.2 million hectares in the studied region (Götaland and Svealand). Thus, the sum of the weight factors for all the samples represents the total forested area. To determine whether a variation between the years exists, individual calculations of the years that were sampled were made. The PROFILE model (Sverdrup and Warfvinge 1993) was used to estimate the pH levels in the year 1840 and when a steady state was reached. The mean value of the relative area in each pH interval during the periods of 1983-1985, 1986-1987 and 1993-1995 was calculated. The results of the calculations and the values modelled by PROFILE are presented in Fig. 10.7. As can be seen in Fig. 10.7 a rapid increase occurred in areas with a pH value below 5.0. According to the PROFILE model, the area that was below pH 5.0 in 1840 was 65,000 hectares in size. In

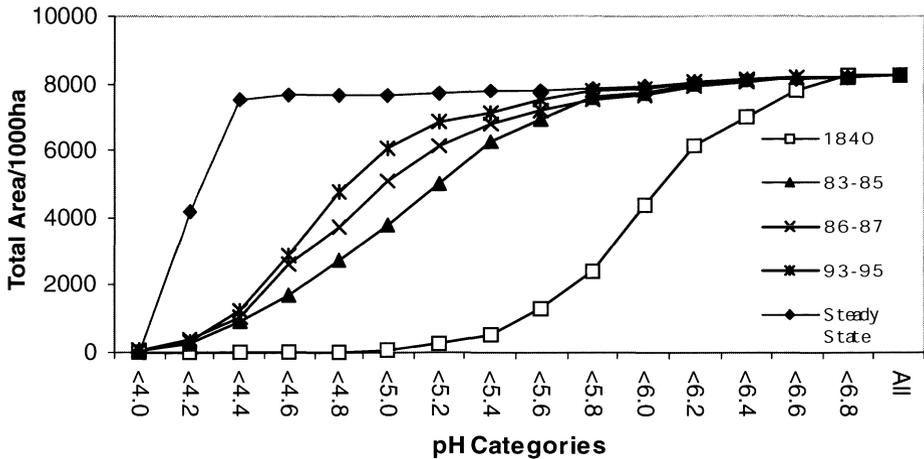


Figure 10.7: Chart showing results from the NFI in southern Sweden. The mean corresponding forested area below a particular pH is shown for 1983-1985, 1986-1987 and 1993-1995. The chart illustrates a continuous trend toward a larger area with a lower pH value in the forest soils of southern Sweden. The pH value in 1840 and when steady state was reached as modelled with the PROFILE model is included as endpoint references.

the years 1983-1985, the figure had increased to 3.8 million hectares and to 6 million hectares in 1993-95. It is predicted by the PROFILE model that when steady state equilibrium has been reached, 7.6 million hectares will be acidified (93% of the total forested area). The time perspective for the system to reach a steady state is dependent upon such factors as the soil characteristics, acid precipitation, biomass harvest, weathering, etc. The costs of restoring the acidified area to a natural pH level would be termed restoration costs. Assume that the liming methods proposed by the National Forest Authority (NFA) lead to a return to a more natural state of acidity of the soil. The cost of acidification can be calculated then by multiplying the area defined as being acidified (below pH 5) by the liming costs per hectare, estimated by NFA to 1,500 SEK ha⁻¹. Thus, the total restoration costs for the acidified soils in southern Sweden amount to some 7,500 million SEK. It should be pointed out that these costs are calculated from the average of all the sampled years. By way of comparison the costs for the areas classified in 1983 and 1995 as being acidified amounted, by use of the same calculations, 5,250 and 9,750 million SEK, respectively. The difference in cost indicates there to have been a rapid increase in cost over time. The annual increase in restoration costs can be calculated in this way as being $(9,750-5,250)/12 = 375$ million SEK.

10.6 Discussion

The amount of forested area defined in this paper as being acidified amounts to 5 million hectares. The annual increase in the size of the acidified area appears very rapid, to judge by the results. It is assumed that the frequency distribution employed is representative of the total area. It is questionable to what extent this assumption corresponds with reality. The soil pH is heterogeneous and varies appreciably within a given area. If the results reflected variations within the material, however, it would be reasonable to assume there to have been only random variation between the years, yet this was not the case, there being a clear trend toward an

increasing number of sites of lower pH. To some extent, the decrease in pH might be explained by the fact that the mean age of the forest stands became greater (Skogsstyrelsen 1998). Tamm and Hallbäck (1986) concluded, however, that stand age has little effect on pH at deeper soil horizons. It is thus questionable whether aging is the only factor contributing to the rapid decrease in pH at this soil depth over such a large area. Harvesting and deposition probably contribute to the rapid increase in the size of the acidified area. Sverdrup and Warfvinge (1995) estimate that 80% of Sweden's forests are in the risk of being negatively affected. For southern Sweden, this figure is 100%, which amounts to 8.2 million hectares. Even after the implementation of the 1999 Göteborg Multiprotocol, the figure will remain at 35% for the country as a whole and at 45% for southern Sweden. In addition to an expected decline in growth, an increase in tree mortality as a result of the critical limit being exceeded will enhance the future costs (Sverdrup et al. 1994). Liming the total acidified area (5 million ha) by use of 3-4 tonnes dolomitic lime/woodash per hectare would cost approximately 7,500 million SEK. If liming eliminates a 5% decline in value for more than 20 years, it will be worthwhile for the project to be carried out. Assuming there to be a similar effect on the Total Economic Value (i.e. loss in terms of recreation value, berries, water, etc.) simply increases the extent to which measures of mitigation can be seen as being justified. The effects of acidification are slow, making them difficult to identify. Because of the large number of variables affecting the growth of forests, it is difficult to detect a decline in growth. However, even a slight decrease in the amount of timber produced result in significant costs. Although it tends to be taken for granted, however, that a reduction in growth will mean a loss in value, it is also possible that a slower growth rate will lead to an increase in the quality of the timber and thus the price paid per m³. There is the theoretical possibility, therefore, that the net loss in revenue will be less significant than one might suppose. However, the major part of the annual harvest is sold to the pulp industry, where the price is much less dependent on quality of the wood. Going towards sustainability requires that changes in the surrounding environment be monitored. This can yield predictions for the future. Such predictions, however, are very difficult to perform, since many uncertainties are involved. Despite the difficulties in defining the economic value of non-marketed goods and services, it may be advantageous to attempt to do so since it diverts the attention away from accounting involving simply marketed goods and services. All would agree that these more vague utilities are important to the common welfare in Sweden. How such values should be quantified is a matter of discussion. Economic valuations are considered by many to be a crude way of valuing the invaluable. Indeed this is right. The methodologies have been criticised by many authors and the uncertainties are considerable. The values involved are also affected by various factors, such as environmental accidents, information, education, etc. Nevertheless, the task remains of justifying the amounts of money it is worthwhile to spend for mitigating the impact of forest soil acidification.

10.7 Reflection on the treat from acidification

In the past 30 years, acid deposition was very much the focus of much of Swedish environmental research, and served as the driving engine for most of the soil chemistry modelling that was done. In the first part of the period (1968-1980), surface waters and fish were much at the centre of attention, later it was realized that this was the last and most visible symptom of a long chain of damages in the ecosystems. The effects on forests were given large attention, and in terms of attitude and genuine ideas much was initially invented by Prof. Ulrich in Göttingen. In many countries, the concern for the long term sustainability of forestry under acid deposition became a major issue and concern. It can be seen in the earlier chapters of the book that acidification grips into many the processes in a forest ecosystem in a quite fundamental way. In large parts of central Europe and Scandinavia, acid deposition changed the soil chemistry so much during 120 years, that most soil scientists in this region have no idea about what the natural chemical state of their soil should be, nor do their textbooks give them any help as the

snapshots of soil conditions found there are taken from systems in the middle of a grand change. Most of the processes we think are connected and important for the concept of sustainability are affected by the changes caused by acidification. Thus, critical loads of acidity and nitrogen, reflect an important aspect of sustainability. The critical load is the capacity of a specific system to be able to receive pollution from the outside. Forest management most profoundly affects this capacity. The critical load concept was developed in order to grip the acid deposition with an ecosystem effect-related concept. This connected for the first time ecological effect to measures and allowed the "end of pipe and technology" approaches to be abandoned. Instead of being at the focus of the process, demands on technology now became driven by environmental quality demands defined by ecological parameters. This was a total paradigm shift in terms of approaching environmental problems. The critical load is a measure for how much the forest ecosystem can absorb acid rain without damage. The concept has been defined and described in detail earlier, and the methodology can be found in Sverdrup and Warfvinge (1988, 1991, 1995) and in Warfvinge and Sverdrup (1990, 1992). Critical loads for Sweden calculated in 2001 are found in Fig. 10.9, based on 1,884 sites from the Swedish Forest Inventory. The maps show the maximum amount of acidity and of nutrient nitrogen that will not cause unacceptable changes to forest ecosystem structure or function. The critical load of acidity incorporate both acidity accompanied with sulphur deposition and with nitrate and ammonium. In the optimization calculations preparing for the protocols, the sulphur and nitrogen deposition is permitted to vary in such a way that they together make up the critical load. The calculations differentiate between soils with different mineralogy, texture and soil moisture, as well as it considers full nutrient circulation and the differences between rooting depth and susceptibility of roots to acid soil conditions. The Swedish Forest Inventory and the Forestry School at Uppsala was most helpful in preparing input data and assist the authors with the actual calculation. Most of the work with critical loads took place at Lund University. The treat is best shown by the exceedance maps, expressed as the 50-percentile, representing the average overload, and the 95-percentile, representing the overload if we want to protect 95% of all sites. For production in general, the 50-percentile maps will be most relevant. Be aware of the fact that the exceedance maps do not show the present situation in the forest at present, but only after a protocol has been fully implemented. This process normally takes several years, if not a decade, and only after that can the soil processes slowly allow the soils to recover. The recovery process is even slower than the political process and has been estimated to take from 20 to 250 years depending on site properties and soil chemistry state at the time of turning the deposition trend. In 1988 the exceedances were very large, the models predicted that a continuation of such deposition levels would lead to large scale damages to soil chemical state (base saturation, pH) and probably to growth damage of a same degree as in Germany and in the Black Triangle (Fig. 10.10). Damages would run into 1000 millions of Swedish kronor per year. The predictions were far from pleasing, but as the calculations were made with the "best available knowledge and data" approach, it had to be taken seriously. It was pointed out, quite rightly, that there were large uncertainties, but nothing of that changed the order of magnitude of the problem in any significant way. Thus, the situation was estimated to involve very serious risks for the future productivity of Swedish forestry, and large efforts were put into the negotiation process under the LRTAP-convention, in fact this work already started in 1987 and is still going on. In 1991, the second sulphur protocol, the "Oslo-protocol" was signed, for the first time based on the critical loads estimated for a number of European countries. It reduced sulphur emissions by 60% from the reference year (1985). This was a great achievement, but still far from what was required to remove the large scale threat against productivity in Swedish forestry (Fig. 10.11). What was most important with the Oslo protocol was that it established a new principle; "effect-based mitigation of pollution", and "mitigation at the source". Both were more important than the actual protocol itself, as this paved the way for the next protocol. In 1997-2000 a new assessment was made and the databases revisited and updated. The soil database underwent significant improvements and during 2000 new results from the SUFOR project was brought in to improve the way critical uptake was calculated. Fig. 10.12 shows the situation in the

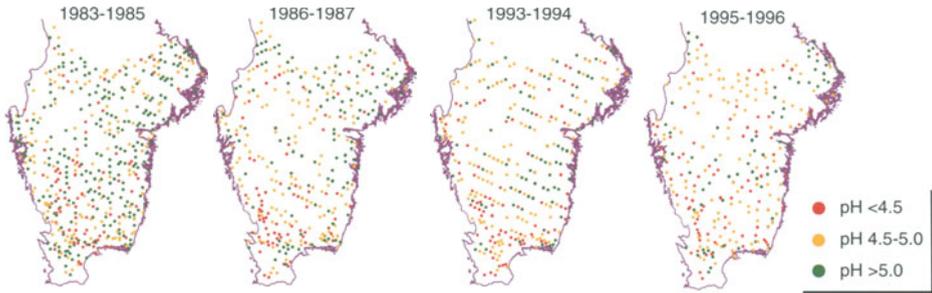


Figure 10.8: *These maps show pH samples obtained in the B-horizon in southern Sweden the period from 1983 to 1996. Each colour represents the pH class given in the legend.*

end of 2000/beginning of 2001. Exceedance is approximately 25-30% of the area (This is at present input data dependent and the true field exceedance is probably 35-45%). Some areas in the south still has significant amounts of exceedance, and here continued forest damage will be seen. In this region, soil pH values continued to sink in a consistent pattern until the print of this book. The work was continued, and in 1999, sulphur, nitric oxides and VOC emissions were linked in a multiple pollutant protocol, the Göteborg protocol. This was a very large step towards compliance with no exceedance of critical loads, and when the protocol comes into full effect much of the threat to Swedish forest productivity will be greatly reduced. In Sweden, we still have significant exceedance with respect to protecting 95% of the sites under the protocol, but a large part of the territory is completely protected, approximately 60%, (Fig. 10.13). There is exceedance in approximately 15-20% of the area (Considering scaling effects, the true field exceedance is probably 25-30% of the area under the Göteborg protocol) and only a small part has large exceedance. Much of the remaining acidity exceedance is now caused by nitrogen deposition, a large part of this is ammonium deposition with large potential for acidification. A revision of the Göteborg protocol is foreseen for the years 2004-2005. The protocol has not yet been fully signed and ratified by all UN/ECE-LRTAP parties to the convention. Thus the protocol will not come into full effect until this has been achieved. Thus, signing of a protocol does not at all imply that the conditions intended in the protocol will become reality until many years/decades later. Likely is that we will approach compliance with the protocol towards 2010 and recovery make take several decades after that date. Until this time good arguments must be brought to bear to bring the concept of sustainable forestry into the critical loads concept. For the Scandinavian countries, sustainability of forestry is both a necessity and an insurance for future prosperity. Forest managers in the field and small forest owners in Southern Sweden could see the effects of environmental problems on their own properties every day, and they simply could not afford to ignore such an evident phenomenon the way an academic institution could and sometimes would do. Not all grasped the urgency of achieving success with the European negotiations under the UN/ECE LRTAP convention in the 1980's, neither was it afterwards fully grasped how close we came to have significant problems, nor how narrowly we escaped from just that threat, and that only with relatively modest damages as compared to what could have been. Had there been no negotiation successes, no protocols, then we would still be looking at "doom-and-gloom" scenarios, and all the field evidence point towards that continued deposition at 1980 level would have created massive damages and widespread crisis in our forests. The negotiating the development of three subsequently improved UN/ECE-LRTAP protocols changed all that, and because of this, the "doom-and-gloom" scenarios will probably never come to be in our Swedish forests. After the Göteborg multi-pollutant protocol 1999, acid rain still remains as a problem for Swedish forestry, but it is no longer represents a large threat to production.

This implies that acid deposition will still cause extra costs for nutrient additions in sustainability management in 40% of our forested area (100,000 km² out of 230,000 km²), and it will probably be able to adversely affect growth and vitality noticeably in approximately 5-10% (20,000 km²), mostly in southern Sweden. Literally speaking, the remaining acid deposition is still "picking money out of our pockets". It may appear not to be much, but everything should be put into its right proportions, Swedens remaining area to be significantly affected by acid deposition and have exceedance of critical loads is still larger than the total forested area of Denmark, Netherlands and Belgium combined.

The present critical loads were set using fairly crude indicators of damage (root dieback/severe needle loss as correlated to soil chemistry parameters). All the available data suggests that this is a kind of last barricade approach, and that the limits should have been set much stricter to protect more subtle and sensitive subcomponents of the forest ecosystem (microfauna, diversity of the vascular plant flora, abundance of red-listed species, biodiversity, seed germination vitality etc.). When damage has developed so far that roots of large trees start to die and significant amounts of needles are lost, then other less observable mechanisms and components have most probably taken a lot of damage. Experimental data (Sverdrup and Warfvinge 1993) also suggest that older trees are far more sensitive than younger, and we need to remember that the limits are often set based on seedling or young plant bioassays. Thus, present limits and critical loads are adapted to the strongest and not the weakest, equivalent to a high risk profile. We may try to rework the definition of critical loads, in to a general principle for any impact caused by man:

"The critical load of a anthropogenic disturbance is the maximum amount of impact from human activity that will cause the maximum acceptable damage to ecosystem structure and function"

The critical issues in this definition will be what we define to be "impact" and how we chose to delineate "ecosystem". Here we should perhaps be more inventive that in the past, eventually, the earth is an ecosystem with humans, infrastructures and all the components of what we traditionally call "nature". This concept will be applicable and flexible if we define load of impact as "acid deposition", "estuary effluent of dissolved phosphorus", "deposition of heavy metals from fertilizers", "amount of stem harvest based forest management", "amount of clear-cut-based forest management", "number of tourists using the forest for barbecue parties", "miles of drainpipes" and so on. It would provide us with a definition for building models and setting limits to any activity that interferes with our ecosystems.

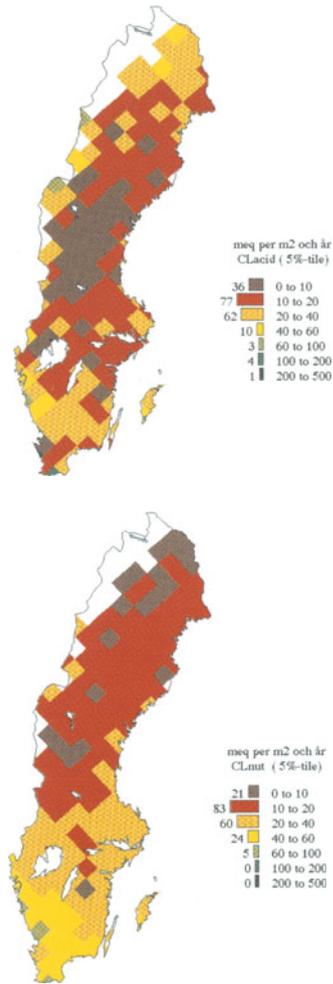


Figure 10.9: The critical load for Swedish forests represents the maximum acid deposition that will not cause damage to growth and vitality. The nutrient critical load applies to nitrogen and is the maximum load that will not cause nutrient imbalance.

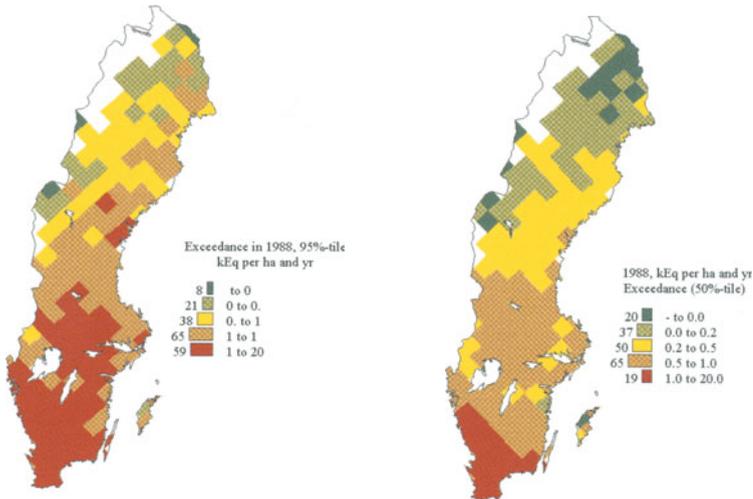


Figure 10.10: *The degree of the exceedance in 1988 was quite severe, and the threat to growth and production potential of utmost significant. 5- and 50-percentile.*

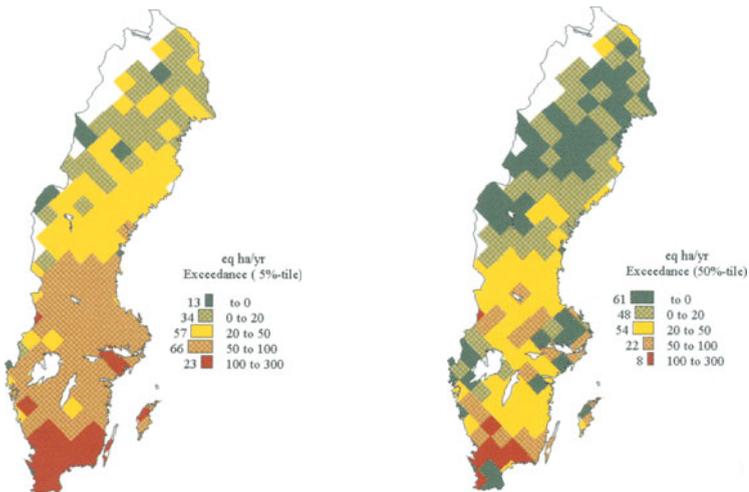


Figure 10.11: *The Oslo protocol improved the situation significantly, even if it was far from what was really needed. 5- and 50-percentile.*

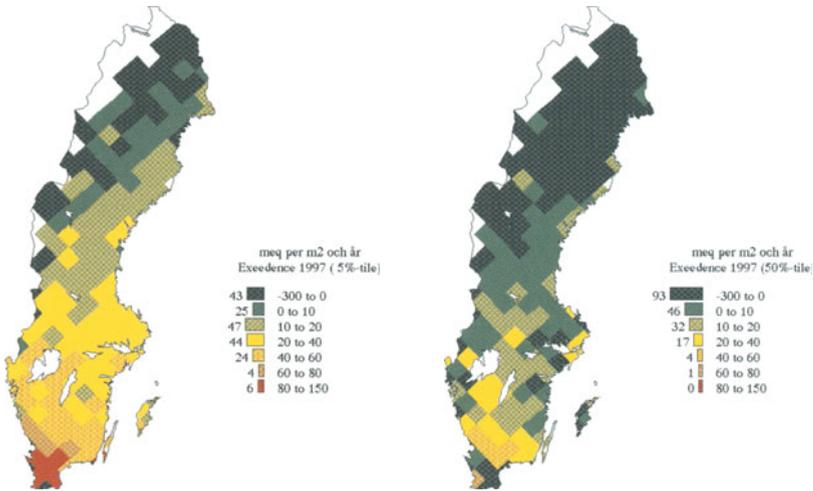


Figure 10.12: The degree of the exceedance in 1997-2000. By this time the situation had improved much, and disaster was no longer looming at the horizon. 5- and 50-percentile.

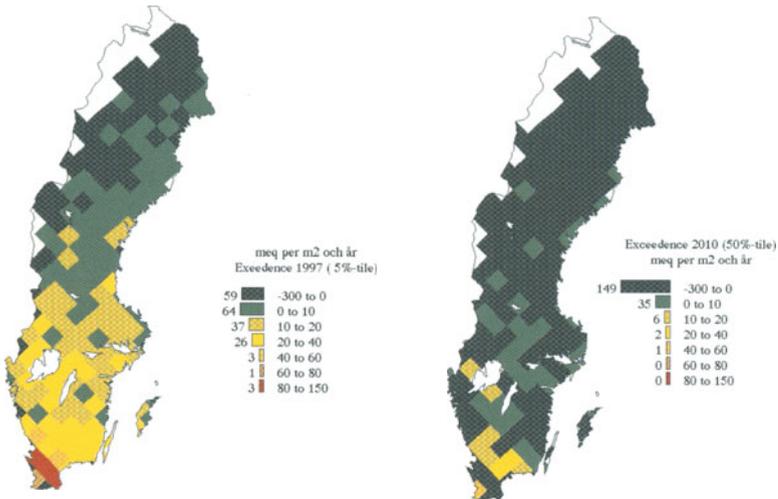


Figure 10.13: The degree of the exceedancen according to the Göteborg multiprotocol. The maps show that the threat from acidification, has to a large degree been removed for most of the country. 5- and 50-percentile.

Chapter 11

Assessing effects of wildlife on forestry

11.1 Browsing damage to forests vs. recreational values

By Jonas Bergquist and Christer Kalén

11.1.1 Introduction

The populations of moose and roe deer increased dramatically in Sweden during the latter half of the 20th century (Hörnberg 1995; Cederlund and Markgren 1987; Cederlund and Liberg 1995). A significant increase in the cervine population was also observed in most other European countries during this period. The regulation of hunting, the decline in predators and changes in land use are believed to be the major causes of this population increase (Kuiters et al. 1996). Both moose and roe deer feed (browse) on young trees and can thus inflict damages on them and reduce the economic yield for the landowners (Gill 1992 ab). Although landowners can derive some benefits from game hunting, they show little tolerance for browsing damages. Most deer species are selective feeders, utilising and only certain plant species for feeding purposes (Hofmann 1989). Moose and roe deer prefer to feed on certain species of shrubs, on young trees and on herbs but avoid grasses (Cederlund et al. 1980). Forest management has a strong influence on the species composition and abundance of ground vegetation, and thus has a variety of possible alternatives influencing deer forage production to selecting between (Hannerz and Hånell 1993; Bergquist et al. 1999), although forestry today seldom utilises the selection opportunities this provides.

Swedish hunters often claim that the availability of forage (food) is decreasing and that this is the cause of increased damages and problems associated with them. They argue, therefore, not for a lowering the cervine population, but for an increase in the availability of food for them in the forests so as to reduce damages in this way. Many foresters are also hunters, making them ambivalent on the matter. A socioeconomic analysis requires the costs inflicted on forestry to be compared with the benefits derived by hunters and by the general public. Theoretically, this allows the optimal levels in socioeconomic terms, both for the herbivore population and for the forest damages that result to be derived.

11.1.2 Browsing damages

Effects on forestry

Large mammalian herbivores such as roe deer and moose, can damage trees by browsing (feeding on shoots and foliar parts), eating bark, fraying or rubbing trees with antlers and trampling on or scratching them. Browsing is generally regarded as the most serious and common type of damage done, although of other types may be more important locally in some cases (Gill

1992ab). In order to simplify the discussion here we use the term browsing to designate all types of damage that moose and roe deer inflict on seedlings and trees. Browsing not only reduces the volume increment, but also the quality of timber in case of heavy browsing (Gill 1992b; Welch et al. 1992). Extensive browsing pressure also increases tree mortality (Gill 1992b; Edenius et al. 1995) since trees are generally able to readily repair browsing damages, browsing does not always result in serious biological or economic damage to trees (Heikkilä and Löyttyniemi 1992; Canham et al. 1994). Because of browsing damages, Norway spruce (*Picea abies* L. Karst) is favoured over Scots pine (*Pinus silvestris* L.) and most deciduous trees due to lower feeding preferences for spruce among roe deer and moose (Hörnberg 1995; Kullberg and Bergström 2001). Spruce appears to repair the loss of above ground biomass better than pine does (Långström and Hellqvist 1989, Lodin 1999). In planting new forests, managers generally prefer spruce because of its lower susceptibility to browsing damages (Lundén 1996). All these mechanisms result in there being higher proportions of spruce in forests than desired. Although to most people, it seems obvious that the level of browsing damages is positively correlated with the density of browsing animals, which has also been shown in many scientific studies (Welch et al. 1991; Tilghman 1989; Bergström and Vikberg 1992), in some studies no effect of the density of browsers on the damage done has been found (Bergquist and Örlander 1998) or more damage has been found to occur at sites of lower browser density (De Jong et al. 1995; Reimoser and Gossow 1996; Bergquist and Örlander 1998). Several explanations of the discrepancy of these results can be suggested. For one thing, the population density of browsers may not have been related to the quantity and quality of edible vegetation available or the area might simply have been too small to study the effects of browser density adequately. Also, it is possible that certain plant species are so attractive that they become severely damaged even under conditions of relatively low browser density (Guthörl 1994). In addition, when browsing damages are studied over large areas, they often correlate well with population density (Brandner et al. 1990; Hörnberg 1995). Home ranges of at least 500 ha for moose and of 100 ha for roe deer have been reported in southern Sweden (Bergström 1996). In both species, however, younger animals frequently disperse and colonise new areas (Wahlström and Kjellander 1995). Due to this dispersing behaviour, young moose or roe deer often emigrate from areas of lower feeding quality, normally displacing a moose that has been shot or roe deer in an area of abundant forage. Thus, to regulate the population density of browsers effectively through hunting (and reduce insofar as possible damages to forestry), coordination of actions over a much larger area than that of a typical private forest estate in Sweden (ca 30-50 ha) is needed. Although no absolute figures can be given, Wahlström (1996) suggested that this can be an area of at least 5,000 ha in the case of roe deer.

Forage availability

An increased in the availability of forage is generally believed to decrease the level of browsing damages, provided the deer population is constant. Several studies appear to confirm this hypothesis (De Jong et al. 1995; Vivås and Saether 1987; Andrén and Angelstam 1993), although there are also studies demonstrating no clear relation between forage availability and damage (Danell et al. 1991a,b; Härkönen 1998). Hypothetically, damage might even increase with availability of forage if more deer were attracted to the stand, although we have not found any cases of this sort in the literature. To regulate damages connected with the availability of forage and ensure positive effects, we probably need to operate over areas much larger than at the stand level. Again, no absolute figures can be given. Site productivity influences both the quantity and quality of forage for browsing animals. Areas high in productivity can support higher herbivore densities. In some studies, damage has been found to increase with productivity, whereas in others it has been found to be reduced or no effect at all has been found (see discussion in Gill 1992a). The mechanism behind such effects is not clear. Important factors other than that of access to alternative food sources may also influence the damage level. Plants on productive sites tend to be more vigorous (healthier or faster growing) than plants growing on poorer sites

(Danell et al. 1991b). Vigorous plants generally repair damage faster (Edenius et al. 1995), grow out of the browsing range more quickly (Guorley et al. 1990), and seem to be preferred as food to a greater extent than plants low in vigour (Danell et al. 1991b, Bergquist and Örlander 1998), making it difficult to determine the relative importance and net results of these effects.

Available forage is not the only factor that attracts deer to an area. For instance, the size and numbers of edge zones between stands able to provide cover and open areas is frequently reported to be an important factor in determining damage (Thirgood and Staines 1989; Kay 1995; Reimoser and Gossow 1996). In a longer time perspective, an increase in the availability of forage also increases the landscape's potential carrying capacity for browsing animals. If there are no controlling mechanisms, the numbers of moose and roe deer increase, and very likely browsing damages as well. Thus, we assume there to be some kind of mechanism that effectively controls the density of moose and roe deer, hunting being one such mechanism.

Estimating the costs of browsing damages

There is an obvious problem in estimating the costs of browsing damages since it is mainly young stands that are damaged and the main costs of such damage are first known when the stand is harvested, which in southern Sweden means after 50 years or more. In forest economic calculations, the incomes and costs (during a rotation) that come about are derived from what incomes and other costs (e.g. timber, labour, machinery, chemicals, and seedlings) are today. The final sum is estimated by discounting the incomes and other costs from the time when they are expected to arise to today's prices by use of a given interest rate (in Sweden normally 3%/calculated for the entire rotation and different management varieties be compared. The reduction in income due to damages can be estimated by use of this method. For a more detailed account of the economic calculations used in forestry, see Cabbage et al. (1991). Knowledge of mortality, growth reduction and the development of defects due to browsing are needed for these calculations. Such data is available for spruce (Mitscherlich and Weise 1982) and for pine (Heikkilä and Löyttyniemi 1992), but for many deciduous tree species such information is scarce. Heavy browsing damages often result in a shift from pine or deciduous trees to spruce. If a site is not particular suitable for spruce, cost may be incurred through production being reduced, which needs to be taken into account in calculating the costs of browsing damages. The reduction in the proportion of deciduous forests, which already today has become only a minor constituent in Swedish forests, threatens many values and opportunities. A diversification of tree species can have advantages for forestry since it is difficult to foresee future markets, the effects of climate change, the degree of soil acidification and various unexpected events. Deciduous forests also possess many "soft" values such as recreation and biodiversity (Drakenberg et al. 1991). Potential threats costs and gains of various types are troublesome to estimate.

11.1.3 Influence on forage availability

Feeding preferences of moose and roe deer

Moose and roe deer have been classified, in terms of their selectivity in feeding, as selectors or browsers. This means that their digestive systems are adapted to utilising plant biomass of relatively high quality (Hofmann 1989). Accordingly they vary their feeding on different plant species and plant parts in the course of a year. In the winter, their diet is dominated by twigs from conifers (mostly pine), shrubs of various kinds and deciduous trees, whereas in the summer they utilise mostly the foliage of deciduous trees and shrubs as well as with herbs as forage. Within these main groups of plant species, moose and roe deer show strong preferences for particular species (Gill 1992a, Hörnberg 1995). Moose are oriented more strongly to feeding on young trees, whereas the smaller roe deer feed more on the field layer. The vegetation in young stands in Sweden is dominated by grass (Bergquist et al. 1999). This group of plant species is more or less avoided by moose and is only used in a limited way by roe deer, mainly in late winter and early spring (Cederlund et al. 1980).

Influence by plant succession

Both moose and roe deer are species ecologically adapted to living in forests (Liberg and Wahlström 1995). In a natural forest, they feed mainly of sites in early succession such as are normally created by "catastrophic" events such as forest fires, insect damage, flooding and windfall. At such sites, an abundance of field flora of higher nutritional quality than vegetation under a canopy usually develops within a few years (Oldemeyer 1987, Johnson et al. 1995, Huges and Fahey 1991). In managed forests, such natural disturbances are relatively uncommon, its instead being forest management that produces early successional sites through cutting. Since in most cases, a stand rotation period is shorter than natural succession (Jedrzejewska et al. 1994; Uuttera et al. 1996, Östlund et al. 1997), forestry is likely to produce a higher frequency of such sites and thus an attractive environment for these animals. This at least partly explains the rapid development of the deer population in managed forests. It also means that that one can expect moose and roe deer to prefer feeding on young stands where food is abundant. For instance, faecal pellets of both moose and roe deer are more abundant in young stands than in older ones, indicative of such behaviour. There are differences between the two species however, the smaller roe deer appearing to spend relatively more time on young clear-cuts (0-3 years) and in older stands than moose, which tend to show a strong avoidance of old forests (8-25 years).

In Scotland, red deer (*Cervus elaphus*) and roe deer that coexists display a similar difference in feeding in terms of stand ages, the larger red deer being more often found in somewhat older though still young stands and roe deer in the younger stands (Welch et al. 1990). Although forestry can usually not vary the intensity of cutting so as to produce more adequate deer forage, this factor should be considered in any calculation aimed at balancing of the number of deer against damage by browsing. When the intensity of cutting is changed one can expect there to be a faster reaction on the part of the smaller roe deer and a longer time lag before moose react to this change.

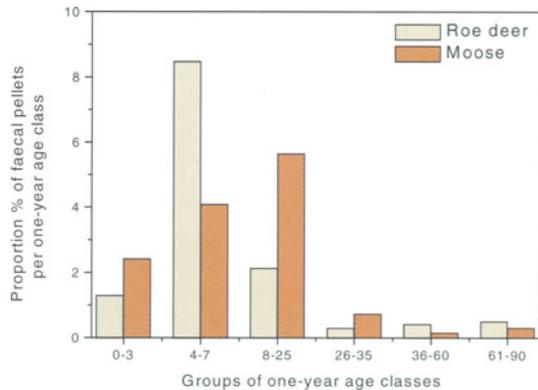


Figure 11.1: *Proportion (%) of faecal pellets of roe deer and moose found in forests of different age classes (years) at the experimental forests in Asa and Tönnersjöheden.*

Influence by stand management

The way a forest is managed can influence both the ground vegetation and the production of forage. In harvesting a stand, the forester can either cut the entire forest or leave a proportion of the trees as shelter-wood. The vegetation that develops under shelter-woods in Sweden often contains a higher proportion of woody species such as dwarf shrubs and tree seedlings than is the case on clear-cuts (Hannerz and Hånell 1993, Bergquist and Kullberg 2001). This means a higher production of winter forage on sites with shelter-woods than on clear-cuts. Slash (branches and tops) can be either harvested or retained. Retaining slash leads to a occurrence of deciduous shrubs and large herbs, used as summer forage, whereas removing slash increases the occurrence of dwarf shrubs and tree seedlings (Kardell 1992; Olsson and Staaf 1995; Bergquist et al. 1999). Slash is sometimes burned. In a study by Dyrness (1973) burning was found to

reduce the total plant cover, but changes in the composition of plant species made it difficult to evaluate the effects on forage production. The soil is often scarified prior planting, this often increasing the regeneration of shrubs and tree seedlings and of certain herbs (Granström 1986; Kardell 1992). Thus, soil scarification appears to increase summer and winter forage, the only negative effect being a reduction in edible late successional plant (Keenan et al. 1994; Bergquist and Kullberg 2001). Herbicides are sometimes used, which has immediate and strong negative effect on forage (herbs and deciduous species) (Huss and Olberg-Kalfass 1982, Vreeland et al. 1998), although from a longer perspective the effect may be less obvious (Vreeland et al. 1998). Since the animals involved feed on twigs, both the tree species selected for regeneration and the density of them obviously influence the availability of forage. After the regeneration phase, a number of other measures are taken that influence forage availability. In Sweden, the first thinning is often made after about 10-15 years. Since this reduces the number of trees, it reduces the winter forage for large deer species such as moose (Härkönen 1998). On the other hand, thinning increases the amount of light at ground level, leading to an increase in ground vegetation and subsequently to a larger amount of forage being available for smaller deer species (Pietz et al. 1999). Another important measure is that of fertilisation, which increases both quantity and quality of forage (Ball et al. 2000). Since foliage from old trees is often preferred as food (Swihart and Picone 1998), it can be best to carry out the harvesting of old trees of preferred species during the winter, giving deer the opportunity to feed on foliage from old trees. Forage production can be enhanced by actively directed measures such as the seeding of attractive plant species when treating the soil (Campbell and Evans 1978) or the cutting of deciduous shrubs in such a way that they bushes out (Karlsson and Albrektson 2000). Forestry management measures clearly determine to a considerable degree the availability of forage for deer. Although there is a relative abundance of literature describing in broad terms the effects of forestry on vegetation and on plant species composition, there is an obvious lack of data showing these effects in terms of the quantity and quality of edible biomass available to deer, which makes it difficult to determine and calculate the carrying capacity of a forest area.

Costs for increasing forage production

In forest management, there are obviously a large number of ways in which deer forage production can be influenced. In general, the costs of selecting different forest management approaches can be calculated by use of traditional forest economic methods.

11.1.4 Recreational value

According to harvest statistics provided by the Swedish Hunters Association the mean densities of moose in southern Sweden appear to be in the range of 5-10 per 1000 ha of forest land. The roe deer population densities in the same area have been estimated to be about 100-150 per 1000 ha (Cederlund and Liberg 1995). These high densities lead to considerable hunting activity, about 500,000 hunters (ca 15% of the adult male population) being engaged in it every year. Roughly 100,000 moose and 250,000 roe deer are harvested each year. In 1994 the price for one kilo of moose meat was estimated at 40 SEK (Bernes 1994), the value of roe deer meat probably being about the same. From a mature moose, about 120-300 kg of meat can be obtained, and from a calf about 60-75 kg of it. For a roe deer, the corresponding figure is about 15 kg as the mean of adults and fawns together. Thus, the socio-economic value of moose and roe deer hunting can be calculated to be approximately 650 million SEK annually, for the value of the meat only. In addition to the meat value, hunters spend money on licence fees, rent to landowners and hunting equipment. A number of studies of the recreational value of hunting have been carried out (e.g. Wibe 1994). Many of these endeavours to determine the economic value of it by letting hunters respond to questionnaires. The method of estimating the recreational value by asking people what they would be prepared to pay if the utility (the experience of hunting) had an ordinary market value is termed the Contingent Valuation Method. This method is

useful for dealing with utilities for which there is no obvious market such as taking walks in the forest, recreational fishing, hiking and hunting. A hunter is prepared to pay between 4,600 to 4,900 SEK (1991 year prices) for acquiring a hunting license for the current season (Bernes 1994). The National Institute of Economic Research (Konjunkturinstitutet) in Sweden estimates the socio-economic value of hunting to be 2,000 million SEK per annum (Jämttjärn 1996). The recreational value of moose and roe deer is likely to be dependent on their densities. Although the presence of a larger number of animals raises the value, the relationship between densities and recreational value need not to be linear.

11.1.5 Conclusions

In order to incorporate browsing damages into a socio-economic model of combined forestry and hunting, one needs to determine how browsing damages are related to population density on a relatively large scale. Population densities should be expressed as numbers of animals per amount of edible food. One should express the costs of browsing damage in not only economic but also social and environmental terms. Better data is needed for estimating how forestry influences the forage production and the carrying capacity of a forest area. To obtain an appropriate balance from a socio-economic standpoint the costs inflicted upon forestry need to be compared with the value of game hunting. The theoretically optimal moose density can be determined from the total value of recreational hunting and the total costs inflicted upon forestry. A socio-economically optimal population can be found for which the net benefit is at a maximum. The demand for hunting, or rather its marginal value, is thought to decrease as the moose population increases. At some point, an additional increase of individuals has a very little effect on the total value. The costs, on the other hand, are thought to increase exponentially as the density of browsers increases (Tilghman 1989). The theoretical optimum from a socio-economic standpoint is found when the difference between the recreational value and the browsing costs is at a maximum.

Unfortunately, little is known regarding the relationship between the density and costs of moose. We do know there to be a positive relationship between density and browsing damages over a large scale (Hörnberg 1995). The challenge is to also find the economic loss these damages would result in. Although the estimate of a value of 2,000 million SEK per annum for recreational hunting would seem to be very high, we have little knowledge of how this value changes with the moose density. Estimating a socio-economically optimal moose population is not an easy task since tools to estimate the degree of browsing at a given moose density is lacking. The fact that forestry itself may alleviate current browsing damages to some extent is a further complication. The costs of a given level of utilisation, or of the introduction of new silvicultural practices, are unknown and are difficult to estimate. A starting point could be to develop a mathematical model for summarising current knowledge of moose browsing and of the response to it of trees.

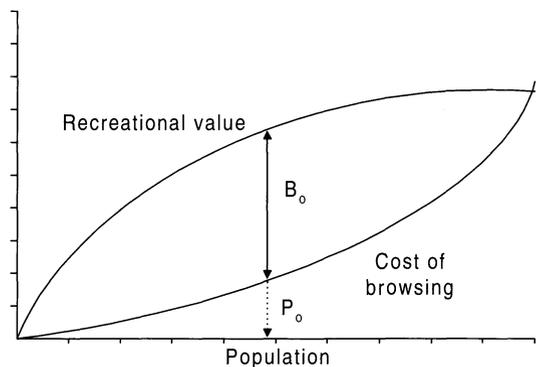


Figure 11.2: A graphical presentation of the theoretically optimal level of moose density from a socioeconomic standpoint. The maximal benefit (distance between the curves) is found at B_o . The optimal level for moose in socio-economic terms density is represented by P_o .

11.2 Dynamic modeling of browsing damages by moose

Christer Kalén and Jonas Bergquist

11.2.1 Introduction

The interaction between forestry and browsing in different deer species (cervids) has escalated during the past century. The density of moose and roe deer represent a relevant issue for any forester today. Although large herbivores inflict substantial cost upon forestry, they also have a considerable value for recreation purposes, implying there to be an optimal level of forest damages versus herbivore density. A model can be a useful tool for analysing the interaction between the herbivore population, available forage and browsing damages. In the present study we consider various approaches to modeling this relationship. Modeling has always been an important tool for explaining or describing complex systems. Today, the term is used in so many different ways that the meaning of the term model has become somewhat blurry. Common to all models, however, is that they represent rough simplifications of a complex reality. One can distinguish at least four categories of models:

1. Conceptual models that aim to visualise a system.
2. Mathematic or theoretical models with the objective of analysing (and simplifying) the mathematical relationship between two or more variables.
3. Empirical or regression models used to present results of field investigations. The driving forces, or underlying mechanisms, are not always of interest. The major disadvantages of this type of modeling are that the mechanistic understanding provided is sometimes poor and that the feedback mechanisms are usually omitted.
4. Dynamic models, which usually take advantage of computer programming in studying a complex system over time, feedbacks being an integrated part of such models. Such mechanistic models are also empirical to an extent, since the fundamental relationships involved at certain stages are based on empirical evidence. A dynamic model may easily become difficult for others to interpret, however.

Generally, a modeling approach can either be top-down or bottom-up. A top-down approach aims at sorting out the main factors influencing a system. In our context, for instance, these are the moose population, the twig biomass per hectare and the average annual twig consumption per moose. By adding variables continuously in this approach, a better resolution or an improved understanding can be obtained. A bottom-up approach starts at a more fundamental level with consideration of basic principles (e.g. optimal foraging theory). Both approaches have their advantages and disadvantages. The top-down approach often simplifies complex dynamic issues, with a considerable risk of being unable to predict with sufficient certainty. Bottom-up models, in turn often fall short, since scaling problems (from the stand to the landscape level) often occur. For a more thorough presentation of the modeling of ecosystems see e.g. May (1976), Ford (1999), Sokal and Rohlf (1995). Various dynamic models of mechanistic character have been developed for studying the interaction between forests and herbivores. Moen et al. (1998) developed a spatial-explicit mechanistic model for the foraging of a single moose in a patch, a model aimed of linking feeding energetics with food availability on an annual basis. Other modeling exercises suggest that the introduction of browsing in forest gap models results in a change of forest structure (Jorritsma et al. 1999; Kienast 1999). We will elaborate on how a model aimed at predicting browse utilisation can be structured and what variables that need to be included. Several modeling methodologies are employed (conceptual, theoretical, empirical and dynamic) for sorting and analysing the dynamics of herbivore feeding and biomass availability. Although we focusing on moose, a similar approach may well be applicable to roe deer, which are of great importance, especially in southern Sweden in particular.

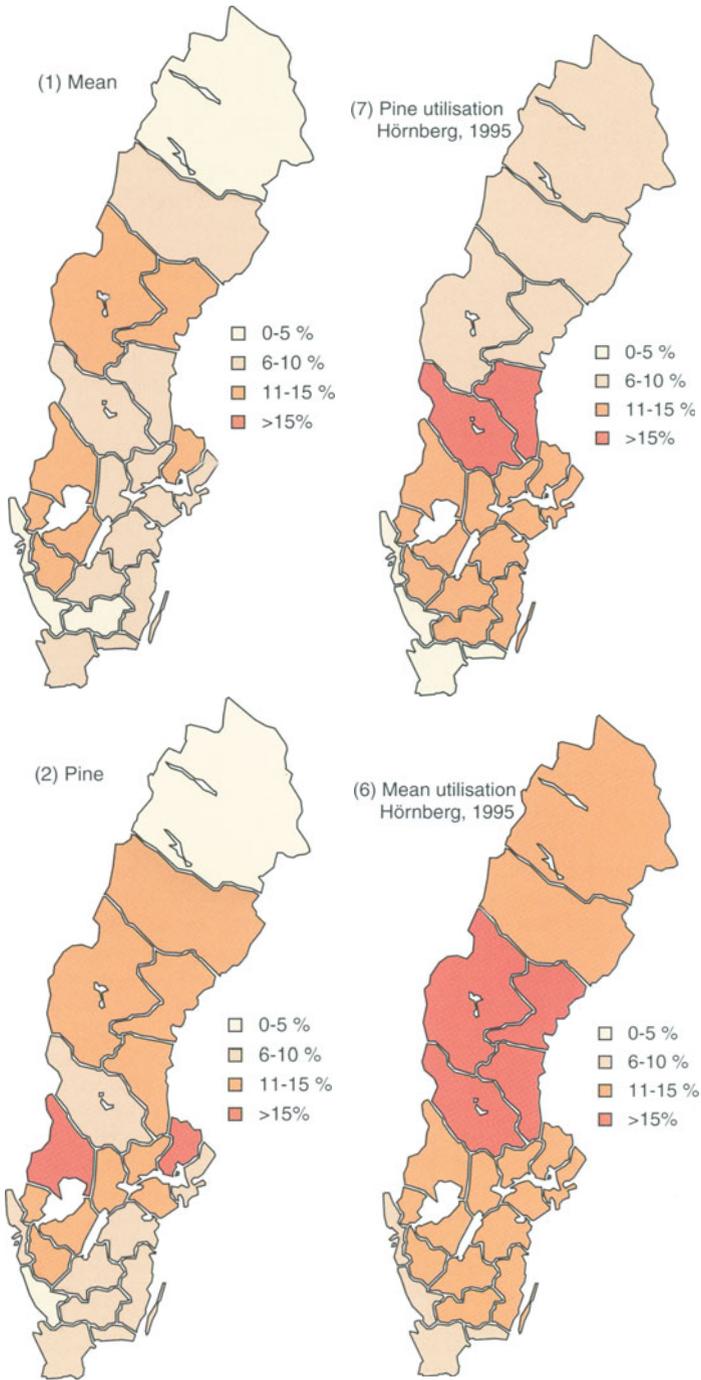


Figure 11.3: *Estimated degree of utilisation of the species consumed (mean 1983-1987)*

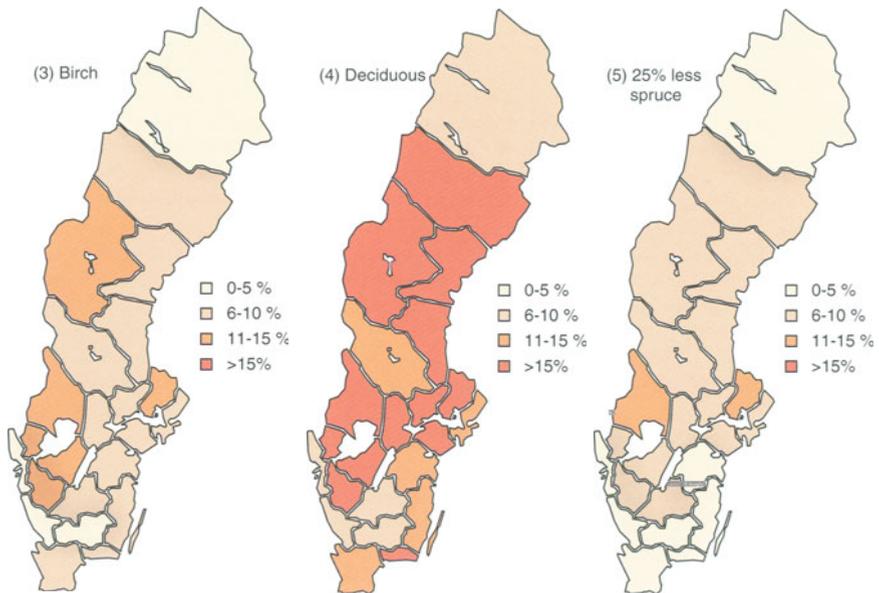


Figure 11.4: *Estimated degree of utilisation of the species consumed (mean 1983-1987)*

11.2.2 Diet selection

Basic foraging constraints

The ability of moose to use food energy resources depends upon the intake of forage, its digestibility and the rate of passage of materials through the digestive tract (Schwartz et al. 1987). Under conditions of abundant forage, two types of factors, physical or physiological, limit the intake rate. At low nutrient values, the physical factors are those of gastrointestinal limitations and passage rates (i.e. bulk limitations). The intake is also regulated by physiological constraints i.e. the ability to utilise the energy content of the food. Thus, when the nutrient value is sufficient to overcome the physical constraints the animal maintains the energy balance by regulating the rate of intake. As the nutrient value in the food increases, the regulation of intake rate changes from the dominance of physical factors towards the dominance of physiological factors. The browsing habits of moose vary during a year. Since food of high nutrient content is more abundant in the summer period, moose gain considerable weight and store it as fat for the upcoming winter. A low availability of forage during the summer can reduce the chances to store sufficient fat ensuring survival during the winter (Anderssen 1991; Schwartz et al 1987). The moose usually suffer from a negative energy balance during winter. The dietary selection of herbivores is best explained at present by optimal foraging theory (Charnov 1976a; Belovsky 1984; Illius and Gordon 1992). One can distinguish three main views regarding the factors behind dietary selection, namely the drive to maximise energy intake rate, the need of essential nutrients, and the avoidance of toxic substances and of substances that reduce digestibility. There is an incentive to be selective of high quality food if resources are abundant (Lundberg and Palo 1993), in line with the marginal value theorem (Charnov 1976b). The idea behind this is that a given resource is utilised only to a certain threshold level before the animal moves on to utilisation of the next resource. Thus, at high forage availability, less biomass is extracted from each tree (Saether and Andersen 1990; Härkönen 1998).

Chemical properties

Although several attempts have been made to determine the preference order of tree species in terms of chemical constituents, no single factor of this type has yet been found (Bergström and Danell, 1987; Saether et al. 1992). The feeding selection of different cervids has been found, for example, by van Wieren (1996) and Niemela et al. (1989) to be related to the availability of digestible organic matter, by Bergström and Danell (1987) to the magnesium and potassium content, and by Bergquist and Örlander (1998) to the nitrogen content of the food. The chemical properties of twigs show a great variation between different tree species (Bergström and Danell 1987), with site conditions and growth characteristics (Bryant et al. 1983), and during the year (Cederlund and Nyström 1981; Faber and Lavsund 1999). The temporal variation in chemical properties and in biomass growth during the growing season coincides with the peak in browsing (Faber and Lavsund 1999). Herbivore feeding has exerted a selection pressure on trees for a chemical defence in buds, internodes and reproductive parts (Davidsson 1993). This chemical defence can be divided in two general groups: carbon-based terpenes and nitrogen-based phenolic resins (Bryant and Kuropat 1980). The function of these defence substances is to reduce the digestibility of the forage and thus the net gain for the browser. It is postulated that, in order to avoid intoxication, the generalist herbivores often undertake a diversification of browse species (Andersen 1991), although the larger herbivores may be less responsive to variation in quality than the smaller, less mobile species (Davidson 1993). The optimal cropping diameter of twigs is dependent on their size and the defence substances they contain. The intake rate increases with the diameter of the cropped twigs (Bergström and Danell 1987; Shipley and Spalinger 1992; Shipley et al. 1999). Twigs of large diameter have a high content of cellulose, however, which is low in energy and is slow to decompose. Consuming twigs of small diameter reduces the intake- and digestion rate, the latter since the concentration of plant defence substances is elevated at the terminal ends of the twigs (Palo et al. 1992). A plant with a lower nutrient content may benefit through a reduced browsing intensity (Lundberg and Åström 1990). Herbivores may compensate for low nutrient content, however, through consuming a larger volume of biomass (Schwartz et al 1987). The availability of resources for growth can affect the chemical defence of plants (Bryant et al 1983). Jia et al. (1997) found that, although moose took larger bites on birches high in phenol content, the total biomass of these trees that they consumed was less. Illius and Gordon (1992) conclude that ruminants such as the moose have a low throughput rate and better digestive efficiency than hindgut fermenters. Studies of the feeding behaviour of moose suggests, however, that their feeding behaviour can be explained less by the content of defence substances than by the amount of edible biomass involved (Niemela and Danell 1988, Edenius 1993, 1994).

Stand and site properties

At the stand and site level there are several factors that can explain dietary selection. The most important factors are height distribution, field layer vegetation, tree species composition and stand density. Trees are most susceptible to being browsed when they are 30-230 cm in height (Faber and Lavsund 1999). Moose have a maximum reach of approximately 250 cm, although trees higher than this can also be utilised since parts of the crown may be within the browsing limit and they may also simply break off the top of a tree. When only the lower parts of the crown are browsed upon, the effect on tree growth is much less, since neither the leading shoot at the top nor the shoots with the highest net rate of photosynthesis are located in this region. As a tree grows higher, it escapes the foraging window. Although the major part of moose forage is obtained from trees, moose also feed on field vegetation. In the summer, field vegetation constitutes about 50% and in winter about 20 % of the moose diet (Cederlund et al. 1980). If the ground is covered by deep snow in the winter, almost no field vegetation can be consumed. In a nationwide study in Sweden, Hörnberg (1995) showed that the abundance of a tree species alone is insufficient to explain its utilisation by moose as forage. Although spruce

is the most abundant species in Sweden, it is utilised to only a low degree (2%), since moose have a specific preference order related to the feeding value of a tree species (Bergström and Danell 1987; Saether et al. 1992; Shipley et al. 1998). The ranking order of different browse species in Sweden is in general as follows: rowan>sallow> aspen>willow>grey alder>silver birch>common juniper>scots pine> pubescent birch»Norway spruce (Shipley et al. 1998). The actual intake is dependent, however, upon the current availability of different tree species (Johnson 1980). It is thus difficult to make a complete general statement regarding the specific diet of moose. Hörnberg (1995) showed that the intake of both birch and pine decreased in the presence of other broad-leaved species, illustrating the principle that the species composition determines the extent to which each species is browsed. As long as browsing continues preferred species may become less abundant which will also have an effect on the preference order (Edenius 1994).

11.2.3 Conceptual modeling of forage selection

Conceptual modeling can be seen as a first attempt to create a dynamic model. In this process, variables can be linked together in a causal loop diagram, providing an overview of the system being studied. The multitude of factors at stand level that determine browsing is overwhelming. Several attempts to sort out important variables have to be undertaken. Although various factors have been investigated in separate studies, no studies have been (or can be) carried out that keep track of all the variables. They can be assembled, however, in a conceptual model. At a tree level, the following model provides insight into the dynamics of dietary selection. According to the model postulated in Fig. 11.5, three major components determine the ranking value of a given tree: availability of biomass, costs of walking, and consumption value. These components, in turn, are determined by abiotic factors such as climate, past events such as browsing and genetic characteristics such as digestibility. Various missing links in the diagram in Fig. 11.5 should be pointed out. Browsed trees sometimes respond with a higher growth rate at the expense of allocation to defence substances being less. However, the diagram can give the impression that browsing results in a higher tree growth rate, which is not the case. Some trees also respond to browsing by an induced defence, resulting in an increase in the concentration of defence substances. To what extent these feedbacks are important for tree behaviour as a whole is difficult to say, although they probably have the potential to explain a certain part of the current behaviour of a tree in terms of past events.

11.2.4 Modelling of intake rate

In 1959, Holling presented a mathematical model of a predator's response to diminishing densities of prey. The model, or equation, is usually known as the disc equation since he used blindfolded assistants to pick up sandpaper discs from the floor. He reasoned in the following way (in considering the moose as the predator feeding on twigs, which are the prey). Assume that a moose has a specific amount of time to search for food. The more time an individual moose can spend on searching, the more food it can acquire. The density of prey also affects the number of prey that can be acquired. At high densities, more prey can be attacked. The efficiency of the moose in finding twigs is also a variable to consider. In mathematical terms, one can sum up these variables in an equation.

$$P_e = t_s \cdot N \cdot a \quad (11.1)$$

where P_e is the number of prey attacked, t_s is the total search time available, N is the prey density and a is the search efficiency, or number of prey found per time unit. Each prey needs to be handled (eaten, digested, etc.). The handling time for a single item of prey is denoted as h . The more prey are attacked, the more time is needed for handling them. Thus, the total handling time (t_h) is:

$$t_h = P_e \cdot h \quad (11.2)$$

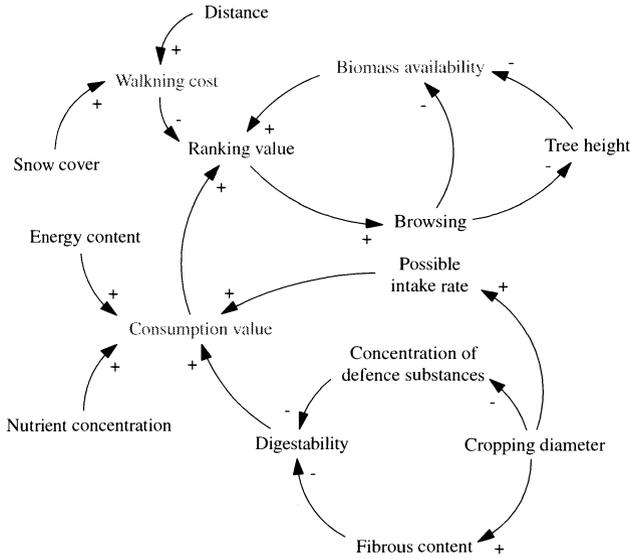


Figure 11.5: A conceptual model clarifying the role of certain variables that determine the likelihood (ranking value) of a given tree being browsed upon. Three factors are being identified as important in determining the ranking value for a given tree. Feedbacks in the system are important, showing how past events can influence behaviour, and also how present events can change future conditions. The signs indicate whether the relationship is positive or a negative (i.e. positive or negative correlation).

or, by substituting P_e by Eq 11.1

$$t_h = t_s \cdot N \cdot a \cdot h \tag{11.3}$$

The intake rate (R) is the total number of prey eaten during the total time for feeding (t), which is the sum of the search time and the handling time. Thus, to calculate the intake rate, the number of prey attacked needs to be divided by the total time available for feeding:

$$R = \frac{P_e}{t} \tag{11.4}$$

which gives:

$$R = \frac{a \cdot N \cdot t_s}{t_s + t_h} \tag{11.5}$$

or when we restructure:

$$R = \frac{a \cdot N}{1 + a \cdot N \cdot h} \tag{11.6}$$

How the intake rate varies with prey density is shown in Fig. 11.6. From this theoretical analysis, it can be concluded that the handling time (h) determines the upper limit to the number of prey that can be eaten per unit of time. The efficiency (a) of a predator determines how rapidly this upper limit is reached as the prey density increases. Lundberg and Danell (1990) suggest that handling time and the search efficiency should not be considered as constants, these instead varying with forage density, so that they should be considered as variables in describing the functional response of moose.

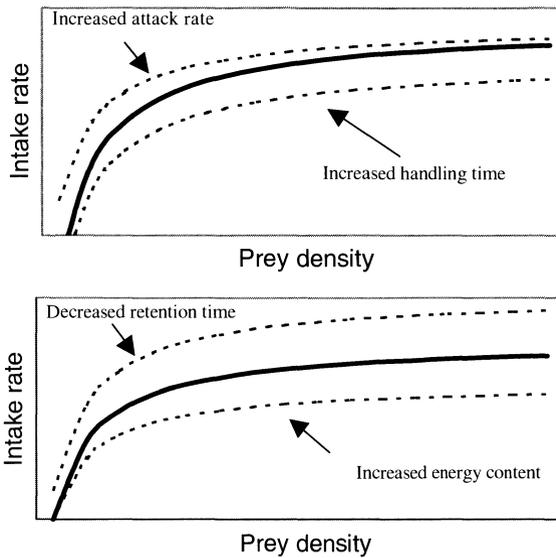


Figure 11.6: As the density of prey increases, the intake rate decelerates towards an upper asymptote. At what rate the upper limit is reached is dependent on the attack rate (efficiency). The level at which the intake rate levels out is dependent on the handling time. The lower diagram shows the effects of an increase in energy content and a decrease in retention time on the intake rate. Both factors affect the consumption rate. As the digestibility increases (a decrease in retention time) the intake rate can become higher. An increase in energy content reduces the intake rate.

Intake rate and dependency of quality

As indicated earlier, moose prefer plants of higher nutrient content. Plants differ in their digestibility, as well, which is dependent upon the concentrations of inhibiting defence substances and of fibre content. To achieve an optimal foraging strategy, an individual should strive to maximise the amount of energy derived per unit of time (which means that the handling time should be minimised). The passage rate is an effect either of higher digestibility or reduction in the levels of substances that partly inhibit digestion. Forage being easily digestible reduces the handling time for it and may thus increase the intake rate. If the nutrient content in the food is high, an individual can meet its needs through consumption of smaller amounts of it. In Holling's disc equation, the handling time is defined quite loosely, a number of different activities connected with foraging such as bite rate, chewing, digestion rate, and gut retention time, being included. Food's being low in digestibility implies the retention time to be longer and thus overall handling time to be longer. Accordingly, animals foraging on food of poor quality have the option of either retaining the food longer in the gut, so as to extract more energy from it or, of allowing the food to pass through the gut quickly, only extracting easily digestible components. Assume further that the amount of energy extracted per time unit affects the handling time. The more energy that is acquired the more time is required to utilise it. Without going further into the physiological functions involved, one can consider the rate of metabolism and the body weight as being determinants of the rate of which energy utilisation. If one then replaces handling time per prey item (h) by the gut retention time (g) and by the energy content

| Species | Occurrence % | Utilisation % | Rank index |
|------------------|-----------------|------------------|------------|
| Sallow/Willow | 6.7 | 23.8 | 87.00 |
| Rowan | 2.7 | 9.4 | 85.03 |
| Aspen | 3.0 | 7.4 | 60.36 |
| Oak | 2.0 | 3.2 | 39.61 |
| Juniper | 6.7 | 10.5 | 38.94 |
| Pine | 11.1 | 16.4 | 36.27 |
| Alder Bucktorn | 1.4 | 2.0 | 35.00 |
| Birch | 20.2 | 23.3 | 28.37 |
| Other (ash) | 0.3 | 0.2 | 15.14 |
| Grey Alder/Beech | 3.6 | 2.0 | 14.10 |
| Spruce | 42.3 | 1.7 | 1.00 |

Table 11.1: *Occurrence, utilisation and ranking index value of different tree species in Sweden. Source: Hörnberg 1995.*

(e) per unit of forage, the equation becomes:

$$R = \frac{a \cdot N}{1 + a \cdot N \cdot g \cdot e} \quad (11.7)$$

These changes affect only the upper limit of the consumption rate, as shown in Fig. 11.6. Various conclusions can be drawn on the basis of this mathematical analysis. First of all, as the digestibility of forage increases (increase in gut rate, g), a greater amount can be consumed per time unit. Secondly, if the energy content (e) of the forage is high, a lesser volume of forage is needed. Placing these general conclusions within the present context, one can note that browsing damages are not simply dependent upon the density of browsers and the amount of forage available, but also the quality of the forage affects the amount of it consumed and thus the browsing damages. Lundberg and Palo (1993), in studying the selection of food with respect to retention time in relation to the quality and abundance of food, suggested there to be a stronger incentive to be selective of high quality food if resources are abundant. To conclude, an increase in the biomass available can reduce browsing damages since: 1) the browsing undertaken is selective, 2) the relative amount derived per tree is reduced, 3) the consumption of high nutritional quality is increased, and 4) a lesser volume of food is needed to fulfil requirement. Although not all of them are consistent, field studies supports these conclusions.

11.2.5 Modeling of browse utilisation on national scale

In this section, a top-down approach to estimating the degree of browse utilisation will be presented. Many variables and processes described in the previous sections are disregarded here since they are difficult to incorporate into a large-scale modeling attempt. The analysis of feeding habits presented in previous sections may be of help to the reader in attaining an understanding of the uncertainty involved in simplifying the complex interactions between a growing forest and a generalist herbivore. The model presented in this section is largely based simple mechanistic principles.

Degree of browse utilisation in Sweden

The degree of browse utilisation is defined here as the relative amount of the twig biomass available that is utilised within a large area of land. Although twigs can be part of the diet

during the summer, the amount consumed is much less than during the winter. We thus simplify twig consumption as occurring only during the winter. Thus, twigs that grow in the summer are assumed to only be consumed during the following winter. The ratio of total annual consumption (C) to annual available biomass (B) is designated as browse utilisation (u). The amount of biomass available during a given year is a function of: the total area of land (a), the twig production per unit of area (p), the relative share of the total area that is suitable for browsing (stands less than 6 meters in height) (k), and the occurrence of tree species suitable for browsing (X). The total consumption is a function of the daily consumption during winter (c), the number of winter days (w), and, number of moose (N). The equation for estimating browse utilisation is:

$$u = \frac{C}{B} = \frac{c \cdot w \cdot N}{\alpha \cdot p \cdot k \cdot X} \quad (11.8)$$

This simple model generates the level of utilisation of the browse available to be expected during the period of a winter. It assumes an equal distribution of the browsing damages exerted on all of the utilised tree species. This simplification may be too crude since we know that the species that are preferred are utilised to a greater degree in comparison to their degree of occurrence whereas others, such as spruce, are largely avoided. There are several ways of incorporating the actual diet involved into the model. Optimal diet theory together with the marginal value theorem do not seem to provide an adequate model for use at a higher scale level (spatial and temporal) since diet selection occurs of the level of the tree and the stand but only a limited extent at the level of the landscape, some alternative being needed for large-scale modeling. The simplest way of estimating the utilisation of different tree species is to use information of species occurrence and species utilisation such as provided by Hörnberg (1995). Through dividing the level of consumption by the occurrence of a given species, relative degree of utilisation of it can be obtained. Creating an index starting from the lowest degree of utilisation provides a ranking list and a ranking value (Table 11.1). To calculate a single constituent of the diet, use is made of the following model:

$$c_i = \frac{c \cdot r_i \cdot t_i}{\sum_{n=1}^n r_i \cdot X_i} \quad (11.9)$$

where c_i represents the relative amount of food item i in the diet, c the daily consumption, r_i the value on the ranking index expressing the preference value for food item i, X_i the relative occurrence of food item i, and n the number of food items present. The tree species included in the diet is divided into four classes: pine, birch, spruce and other deciduous tree species. The degree of utilisation of the different species classes can be estimated using equations 11.8 and 11.9. This allows the browsing damages exerted on species i in an area with a specific composition of tree species to be obtained:

$$u_i = \frac{c \cdot r_i \cdot w \cdot N}{\alpha \cdot p \cdot k \cdot \sum_{n=1}^n r_i \cdot X_i} \quad (11.10)$$

The national forest survey was used to obtain values on k and X. Forest statistics (1998) were used to obtain data on a. Daily consumption c was estimated to 6 kg of twig biomass per day and animal during the winter (Renecker and Hudson 1985). The moose population was estimated on the basis of figures given by Hörnberg (1995) and in Forest Statistics (1998). The number of winter days ranges from approximately 200 in the north to 100 in the south. The production of twig biomass is assumed to be correlated with site-specific productivity (bonitet), which ranges from 2 m³ha⁻¹ in the north to 8 m³ha⁻¹ in the south. Multiplying the productivity by 150 yields a reasonable estimate of twig biomass. The mean occurrence of each species class was obtained for each county. The degree of utilisation was divided into four classes (0-5%, 6-10%, 11-15%, >15%). Estimates were made (Fig. 11.3) for

1. all tree species consumed (mean utilisation)
2. pine

| Age | 1-2 | 2-3 | 3-4 | 4-9 | >9 |
|----------------|------|------|------|------|------|
| Fecundity rate | 0.22 | 0.82 | 0.90 | 1.64 | 1.17 |

Table 11.2: *Fecundity rate of females at different age classes. Sylvén et al. 1987*

3. birch,
4. deciduous trees
5. a scenario of mean utilisation in which 25% of the young spruce stands were replaced by deciduous tree species

Field observations of utilisation derived from the NFI (Hörnberg 1995) are presented Fig. 11.3 by way of reference. Since these observations are for the years 1969-1972, they are not very appropriate for validation. No other data is available, however. On the maps presented in Fig. 11.3 one can note there to be an elevated level of browse utilisation in the central parts of Sweden. This is noticeable in the estimates made in the sampled data obtained from NFI. The model predicts a lower degree of utilisation in the upper north than given by Hörnberg (1995). Värmland and Uppsala have the highest degree of utilisation in terms of the model. Since the data provided by Hörnberg (1995) is divided up into only five regions, thus giving a lower degree of resolution, these patterns are absent in Fig. 11.3. A difference is obvious between the utilisation of different species (pine, birch and the class of deciduous trees), although the overall trend is similar. Replacing 25% of the young spruce stands by deciduous stands results in a lower degree of utilisation. Thus, a large-scale increase in amount of preferred species present leads to a reduction in the browsing pressure, which can result if the herbivore population is being held constant. Although no validation procedure has been undertaken, the results look promising. The model appears to capture the overall pattern found in the observed data. The degree of utilisation also falls within the same order of magnitude as in the field observations.

11.2.6 Predicting moose population and degree of utilisation

Population Dynamics

The moose, *Alces alces*, is the largest deer species in Sweden, up to 2 meter tall at shoulder-height. A mature bull weighs about 500-700 kg. The biological age for a moose is one of about 20 years. The mating season occurs in September and continues until the beginning of October. The cows are pregnant for 230-240 days and usually deliver 1-2 calves. They weigh about 10-17 kg at birth and gain approximately 1.5 kg per day. The calves suckle for three to five months. By October, the calves reach a weight of about 150 kg. During the first year, they are dependent on their mother for finding forage. As in any animal population, the moose population is regulated mainly by the number of births and number of deaths per time unit. The number of births is dependent on the number of females and the number of offspring per female (fecundity). Table 11.2 shows the mean fecundity at different ages for female moose in Sweden. The mortality is mainly the result of predation, starvation, disease, senescence and hunting. The size of the population affects how many animals that harvested, die of starvation or fall under the claws of a predator. Since hunting is the major cause of death of moose in Sweden, the effects of predation and of starvation will be disregarded. A population model was constructed having the basic structure and data input presented by Sylvén et al. (1987). Various features that have been added are described later on. The model consists of 14 cohorts describing the life span of a population. The birth rate is calculated from the number of mature females and the fecundity at a specific age. The fecundity rates given in Tab. 11.2 were employed. At high relative densities, the relative fecundity decreases, due to the availability of forage being lowered (Sand

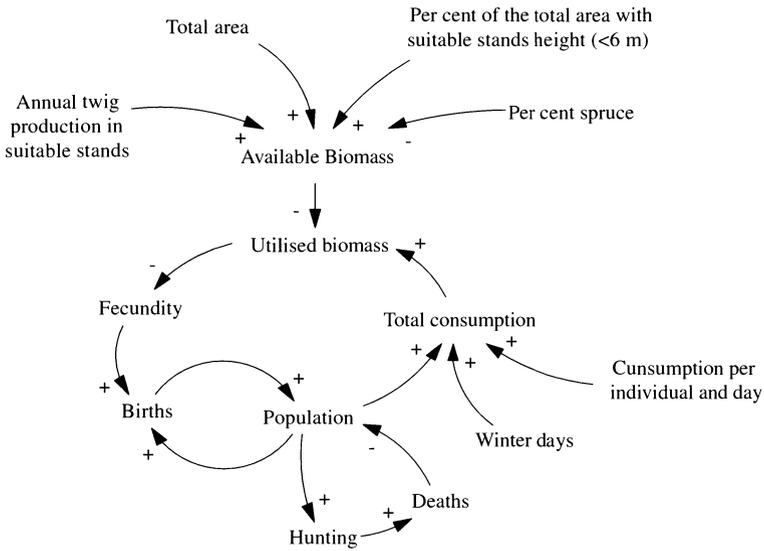


Figure 11.7: A simplified diagram of the model of population dynamics.

et al. 1996; Saether et al. 1996; Wallin et al. 1995). We estimated an arbitrary sigmoid curve as representing the link between the available forage per individual and the relative fecundity. When the utilisation of shoot biomass increases to above 40%, the fecundity rate declines. The basic structure of the model is shown in Fig. 11.7.

Hunting

The moose population in Sweden is regulated through use of a license system that controls the hunting intensity. A fixed number of moose is allowed for harvest within a given hunting area each year. In addition, the period for hunting is regulated by the authorities, being in the autumn and early winter. It is possible, through use of these management restrictions, to regulate the number, age distribution and sex ratio of the moose population. Setting an annual harvesting strategy in terms of an exact number of individuals makes the model very sensitive to over-harvesting. A goal of a population of given size (or density) is more appropriate and better reflects reality. In the model, the annual harvest is thus calculated from a preferred population size. So as to be able to adjust the level of harvested calves and yearlings, relative to the total harvest, one can choose a particular hunting pressure for calves and yearlings. Different harvesting strategies can be evaluated in terms of harvested biomass (kg of meat) and number of harvested animals.

Browse utilisation

By varying the available forage per hectare, one can study the effects of browse utilisation (i.e. browsing damages). Equation 6 is employed to this end, being parameterised as follows. The density of moose is held constant by hunting (10 per 1,000 ha). The number of winter days in southern Sweden is estimated to be 100. Sixteen percent of the total forest area in southern Sweden is below 6 m in height (national forest survey). The species that are consumed (non spruce stands) occupy approximately 50 percent of this. The production of shoot biomass in

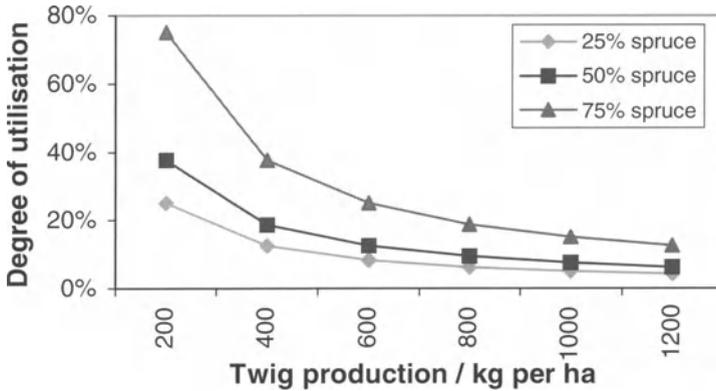


Figure 11.8: By increasing the percentage of spruce stands in a landscape the degree of utilisation of them is increased if the moose population is held at constant density (10 per 1,000 ha).

these stands is assumed to be 1,000 kg per ha. The daily consumption of shoot biomass for a moose in wintertime is estimated to be 6 kg. An optimally foraging moose only forages on a particular prey (plant species) if doing so is more advantageous than searching for a prey of greater nutritional value (Charnov 1976a, Belovsky 1984).

Since spruce is the least preferred tree species in this respect in Sweden (Hörnberg 1995), it is in the model avoided as long as there other species are available. Increasing the relative amount of spruce stands within the landscape leads to an increase in the browsing pressure on the more preferred species (Fig. 11.8). Although at some point spruce would also be included in the diet, this is not considered in the model, mainly because other food sources need to be exhausted to a very high degree before spruce becomes a significant constituent of the diet. Letting the value of each parameter in equation 11.8 be drawn randomly from a selected distribution allows a simple uncertainty analysis to be performed. Since for each parameter the distribution is unknown, a uniform distribution is employed, a random number between specific intervals being drawn prior to the model being run. The parameters are distributed uniformly, with a deviation of 10% from a central value. A series of 100 runs were carried out and mean and standard error of utilised biomass was calculated. The procedure was repeated for three different moose densities (5, 10 and 15 per 1000 ha) (Fig. 11.9). The results of the uncertainty analysis suggest that browse utilisation increases from approximately 3% to 12% at 5 and 15 animals per 1,000 ha, respectively. In Fig. 11.10 it is also evident that the variation in browse utilisation increases with moose density. We are unable, at this stage to determine the damages inflicted on specific tree species and at particular sites. The uncertainty analysis suggests, however, that including

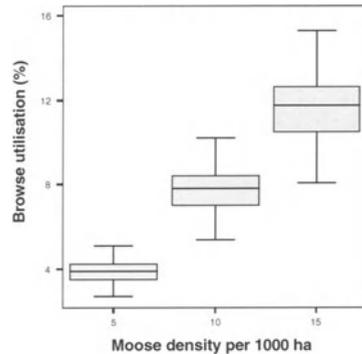


Figure 11.9: The box plot shows the results of the uncertainty analysis. The average percent of utilisation of the browse available was 3.9 (SD 0.5), 7.8 (SD 1.1) and 11.7 (SD 1.6) at low, medium and high densities, respectively.

an uncertainty of 10% in the parameters has an effect on the browsing damages that occurs. Thus, estimating browsing damages with accuracy depends on more than simply knowledge of diet selection and of consumption levels. Accuracy of the data concerning the about number of winter days, the moose density and the stand characteristics is of importance for one to be able to make meaningful statements regarding the degree of browsing damages.

11.2.7 Conclusion

Four different methods were employed for analysing browsing damages: a conceptual, a theoretical, an empirical and a dynamic method. All of these have their advantages and disadvantages. The assumptions made in any of these have an impact on the conclusions that can be drawn. Although the construction of models is a slow iterative and process, it results in an increased understanding of the complexity and the dynamics of forest-herbivore interactions. A general conclusion one can draw is that browsing damages are dependent on several factors, the herbivore density and the area of habitat which is suitable being the most important of those. An increase in tree species (such as spruce) which the moose avoids, increases the damages to the more preferred trees. The conceptual models reflect numerous factors that influence diet selection at the tree level. This is important since it indicates that diet selection is dynamic, so that it is difficult to state what specific diet that is typical of moose. The diet is highly dependent upon the availability of different food sources at a specific location at a particular time. Describing the moose diet in a generalised way on an annual basis and for a large area may under some circumstances be inappropriate. If one assumes there to be a constant species composition, however, it may be appropriate. Since if one wants to test different strategies, however, such as increasing the amount of deciduous stands, the diet changes and with it the realised ranking index to be used. The biomass production employed in the empirical model is an estimated mean value for stands below 6 meters in height. In reality, the twig biomass increases with the height of the trees, up to a point where the crown escapes the foraging window. A higher growth rate shortens the time the stand remains in this foraging "window". Each tree species also differs in the amount of biomass available, making this generalization quite inexact. In terms of the model, replacing spruce stands by deciduous tree species appears to have a stronger effect in southern parts of Sweden according to the model. A likely reason for this is that the occurrence of spruce increases towards the south (Fig. 11.8), exerting stronger pressure on the more preferred tree species. A number of conclusions can be drawn from Fig. 11.10. Since moose generally avoid spruce, the area of suitable habitat decreases towards the south. Although the availability of suitable stands is highest in northern Sweden, the moose density there is not correspondingly large. A likely reason for this is that there is a gradient of increasing productivity towards the south. In the model, 100% of the forage is found within suitable forest stands (these less than 6 m in height). This is a crude generalization since forage is also found elsewhere. The field layer is not taken into consideration either. In northern Sweden, this has less impact because of the snow cover. In the southern parts of Sweden, it may be relevant since nowadays the snow cover in the winter is sporadic and variable. Judging from data on observed browsing damages (Forest Statistics, 1998), browsing damages have a substantial local effect. Thus, browsing is not spread evenly over the area of suitable stands. Some stands may escape herbivory, whereas others may suffer from strong browsing pressure (above 50%). Fluctuations of this sort are not detectable by use of the present approach. The dynamic model developed in this paper is in fact non-dynamic, the only factor considered as regulating population size of moose being hunting. Since the population never approached the carrying capacity of the habitat, there was no reduction in fecundity. Thus, the dynamics in the model were basically repressed since feedback from forage production or of the population size never became realised. Thus, the model may give the impression that browse utilisation is independent on the dynamic interaction between moose population and stand growth. One reason for this impression is that we assume we can control the population density of moose, although we know little about this. Wallin et al. (1995) showed there to be a rapid increase in the moose population when it was

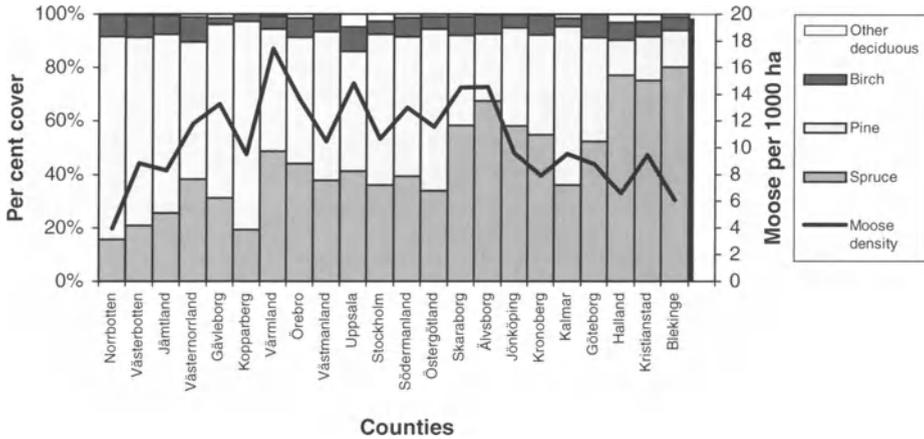


Figure 11.10: *The ratio of young stands of spruce to pine shifts in a north to south direction in Sweden. Young birch stands are quite evenly distributed and occurs to an extent of 5-10%. Young stands of other deciduous tree species are a minor constituent in Swedish forests generally. The line shows the mean density of moose between 1982 and 1992.*

allowed to develop freely. In their study, there was found to be a carrying capacity of well above 50 per 1,000 ha. This indicates that under a "normal" harvest regime, fecundity is not repressed. For the further development of models in the forest-deer arena, emphasis should be placed on simplifying diet selection by use of a mechanistic model. We are not able today to simulate the effects of leaving more preferred tree species in a managed forest stand. Browse availability needs to be examined further in terms of species, height and density parameters. The variation of field vegetation has been disregarded in this paper, which limits the usefulness of the model that is presented. In order for the model to be more useful for the operational management of resources, socio-economic parameters need to also be incorporated into it.

Chapter 12

Sustainability in spruce and mixed-species stands

Assessing nutrient sustainability for single stands at Jämjö

Gunnar Thelin, Harald Sverdrup, Johan Holmqvist, Ulrika Rosengren, Magnus Linden

12.1 Introduction

It is important to find and develop forest management strategies that do not result in the depletion of soil nutrient pools and at the same time are financially competitive in relatively short-term, 1-2 rotations. Mixed-species stands, which combine the elements of fast growing, highly productive softwood and more slowly growing, high quality hardwood, appear to be an interesting alternative in this respect, one worthy of further investigation. Among the mixtures of trees tried out in southern Swedish forestry thus far the Norway spruce/oak-system developed by Erik Stål and his heirs at their estate in Jämjö, Southeastern Sweden, appears promising. According to the owners, the mixed-species stands perform well financially. The management system is well defined and aims at the production of veneer-quality oak wood with Norway spruce as an economic buffer early in the rotations (Ståål 1986). We decided to investigate both the biogeochemical and the economic sustainability of the mixed-species stands there as well as of the adjacent Norway spruce monocultures. Matters of biodiversity are as yet to be investigated. It is reasonable, however, to expect there to be better possibilities for maintaining a high level of biodiversity in mixed spruce/oak forest stands than in Norway spruce monocultures.

12.2 Objectives

In order to compare the nutrient dynamics and the nutrient sustainability of Norway spruce monocultures and mixed-species stands, a field experiment was set up. The experiment had the following objectives:

1. To compare the nutrient dynamics of mixed-species stands and of Norway spruce monocultures during otherwise comparable conditions, using nutrient budgets.
2. To investigate the effect of mixed-species stands on the nutrition, vitality, and production of the trees.
3. To use site data as input to models (PROFILE) for assessing the nutrient sustainability of mixed-species stands as compared with spruce monocultures.

An additional objective was to analyze the economic sustainability of different forest management scenarios.

12.3 Site description

The study site is located at Flakulla, Jämjö in the county of Blekinge in southeastern Sweden (lat. 56° 53', long. 15°16.5', alt. 60 m). For the period of 1961 to 1990 mean annual temperature was 6.9 °C and mean annual precipitation was 550 mm. The bedrock is granitic achaeon, rich in intrusive eruptive rocks. The soil mineralogy is rich in feldspars and dark minerals. The marked presence of epidote can be attributed to the Mien eruptive formation in the vicinity and to the hydrothermal activity associated with it. The soil was found to be of the dystric cambisol type in both the Norway spruce monocultures and the mixed-species stands. This points to the original forest cover several centuries ago having been broadleaf with sparse presence of pine. The soil texture is silty loam. The forest owner's main objective is to produce veneer-quality oak wood in the mixed-species stands. The total area of the farm is approximately 300 hectares.

12.4 Management description

The silvicultural procedure can be described as one of managed succession. When mature oaks are cut, natural regeneration of oak and other species is allowed to occur. Additional seedlings are planted on areas on which natural regeneration of oak and spruce is not sufficient. A few years after regeneration, high quality oak seedlings are selected in the stands at approximately 8 m intervals. The aim is to in the mature stands have high quality oaks at approximately 15 m intervals. In early thinnings spruce is favoured alongside the oak, due to its high volume production. When the oaks reach the age of approximately 30 years, the trees shading them are cut in order to provide the oak crowns maximum light so as to maximize the stem production. The oak stems are pruned up to 8 m. Naturally regenerating spruce, beech, birch, etc. are used to shelter the oak stems from light, limiting the growth of quality-reducing water shoots. The neighbouring trees are cut when they restrict oak crown development. The oaks are cut finally at approximately 120 years of age. The stand development from regeneration to a mature oak-dominated stand can be divided into three phases:

1. A young stand of fast growing birch and other trees as shelter-wood for the oak and spruce.
2. A stand of intermediate age dominated initially by spruce and then with a decreasing proportion of spruce up to approximately 70-80 years, when the last spruce are cut.
3. A mature stand dominated by oak, with stem-sheltering undergrowth.

Although management controls stand development it should be emphasized that the separation into the three phases referred to is a description of the inherent development of the stand rather than of the silvicultural procedures that are followed. The autecology of the different species involved governs the management of the stands.

12.5 Experimental setup and sampling

In 1997, four blocks were set up, each consisting of a 30*30 m plot of Norway spruce monoculture and a 30*30 m plot containing a mixed-species stand of mainly Norway spruce and Oak. The Norway spruce stands were approximately 60 years old. The mixed-species stands have a variable age structure; the age of the dominant oaks range from 40 to 80 years. Diameter at breast height (dbh), total height, and crown length were measured on all of the trees in the mixed stands. In the spruce stands, diameter at breast height was measured on all the trees and height on every fifth tree. A function for expressing the relationship between diameter at breastheight (dbh) and height in the spruce stands was constructed ($r^2=0.8$), and used to calculate the height of the rest of the trees in the stand. To calculate the harvestable wood biomass the functions given below were employed. In the functions for Oak and Beech (Hagberg and Matern, 1975),

| Species | Density | Species | Density |
|---------------|------------------------|----------|------------------------|
| Aspen | 400 kg m ⁻³ | Birch | 600 kg m ⁻³ |
| Mountain ash | 400 kg m ⁻³ | Oak | 690 kg m ⁻³ |
| Norway spruce | 400 kg m ⁻³ | Beech | 720 kg m ⁻³ |
| | | Hornbeam | 750 kg m ⁻³ |

Table 12.1: Wood densities used in the study

V_B was used for stems with a fork below 2/3 of the height of the tree. dbh is diameter at breast height, h the tree height, and k the distance from the ground to the green crown. For oak it is as follows (dm³ ob):

$$V = dbh^2 \cdot (0.02996 \cdot h + 0.008291 \cdot k) + 0.05960 \cdot dbh \cdot h \quad (12.1)$$

$$V_B = dbh^2 \cdot (0.03301 \cdot h + 0.008291 \cdot k) + 0.05960 \cdot dbh \cdot h \quad (12.2)$$

For aspen (dm³ ob) it is

$$V = dbh^2 \cdot (0.01548 + 0.03255 \cdot h - 0.000047 \cdot h^2) - dbh \cdot (0.01333 \cdot h + 0.004859 \cdot h^2) \quad (12.3)$$

For beech (dm³ ob) it is

$$V = dbh^2 \cdot (0.1213 + 0.01721 \cdot h + 0.0003929 \cdot h^2 + 0.005261 \cdot k) + 0.004861 \cdot dbh \cdot h^2 \quad (12.4)$$

$$V_B = dbh^2 \cdot (0.1213 + 0.01483 \cdot h + 0.0003929 \cdot h^2 + 0.005261 \cdot k) + 0.004861 \cdot dbh \cdot h^2 \quad (12.5)$$

For Norway spruce and birch the expressions used by Marklund (1988) were adopted. For Norway spruce it is as follows (kg d.w. ob):

$$\ln d.w. = 7.469 \cdot \frac{dbh}{dbh + 14} + 0.0289 \cdot h + 0.6828 \cdot \ln h - 2.1702 \quad (12.6)$$

For Birch (kg d.w. ob) it is

$$\ln d.w. = 8.2827 \cdot \frac{dbh}{dbh + 7} + 0.0393 \cdot h + 0.5772 \cdot \ln h - 3.5686 \quad (12.7)$$

The beech functions were also used for hornbeam and the birch function was also used for mountain ash. The wood densities (kg m⁻³) used for calculating d.w. from the volume or vice versa are shown in Tab. 12.1. Samples for analysis of the stemwood, the branchwood, and the spruce foliage nutrient concentrations were taken in February 1999. The leaves were sampled in August 1998. In the spruce stands, four randomly selected trees per plot were sampled. For the other species, one randomly selected tree per species and plot was sampled. For spruce, wood cores including the bark were taken at four locations: at breast height, at the bottom of the green crown, at the seventh branch whorl, and at half the distance between the bottom of the green crown and the seventh branch whorl. For the other species, cores were taken at breast height, at the bottom of the green crown, and at half the distance between the bottom of the green crown and the top. Care was taken to not sample damaged or severely suppressed trees or trees with a disturbed growth form. Roots were sampled down to a depth of 50 cm in the mineral soil in two 50*50 cm monoliths per plot, in August 1998 and in May 1999 (Sudhaus 1999). Soil samples were taken in August 1997 and in September 1999 at 20 points along the plot diagonals, using soil corers. The height of the organic layer was measured and the mineral

soil was sampled in four layers, at 0-10 cm, 10-20 cm, 20-30 cm, and 30-50 cm. For quantitative assessment of the mineralogy additional samples were taken in November 1999 from the C or B/C horizons in each plot.

Throughfall collectors, litter traps, and soil lysimeters for the collection of leachates at 50 cm depth in the mineral soil were installed in August 1997. Litterfall was collected and soil water samples were taken on four occasions: in November 1997, March 1998, June 1998, and August 1998. Since the throughfall measurements were seriously disturbed in the winter - the rain collectors were damaged by frost and deer drank the water from the snow collectors - values obtained for the open field deposition and the throughfall at two nearby sites monitored by IVL were used to calibrate the throughfall at Jämjö: Kallgårdsmåla (67 year old spruce) and Glimminge (83 year old oak).

12.6 Chemical composition and nutrient dynamics

In the mixed-species stands, the stem biomass per ha is only 54% of the stem biomass found in the pure Norway spruce stands (Table 12.4). The lower stem biomass in the mixed-species stands means that there is a lower stem nutrient content per ha for most of the nutrients, despite that the stem nutrient concentrations are greater in the deciduous species. N is an exception with higher stem N content per ha in the mixed species stands. At harvest this would result in a proportionally greater removal of N in the mixed species stands than in the pure stands. This implies that maintaining mixed-species stands should be a good strategy in systems close to the N saturation point. The yearly litterfall is approximately the same in the pure and in the mixed-species stands (Tab 12.4), despite that the stem biomass in the pure stands is almost double that in the mixed-species stands. Higher concentrations of N, K, and Mg in the litter of the mixed species stands indicates a higher litter quality, which in turn has a positive effect on the mineralization and on the nutrient availability. There were no differences in soil pH or in base saturation between the pure and the mixed species stands, contrary to what had been expected (Tab 12.8-12.7). Differences in earlier management represent one possible explanation for this. Before planting, 60 years earlier, the areas on which the monocultures now stand were probably arable land and the areas now containing the mixed species stands were pasture land of low intensity. However, the levels of extractable K and Mg appear to be higher in the mixed species stands. More importantly, the CEC is greater in these. This implies the base cation storage capacity to be greater in the mixed species stands than in the pure Norway spruce stands. The higher CEC is probably due to the proportion of organic matter being higher in the soil beneath the mixed species stands. This in turn may be due to a greater rooting depth in the mixed stands and to the populations of burrowing earth worms moving organic material throughout the soil profile there being greater as well. The greater proportion of organic matter may also explain the higher soil concentrations of Al and N in the mixed stands. Note that this had not increased the leaching of N and Al. Instead the composition of the lysimeter water indicates the leaching of Al and nitrate to be greater in the pure Norway spruce stands. The effective present growth was estimated to be $15 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$ in the spruce monocultures and $7.8 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$ in the mixed-species stands that consisted of oak ($5.3 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$), spruce ($2 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$), and other species ($0.5 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$). The rotation average growth was estimated from St¹l (1986) and by the authors to be $10 \text{ m}^3 \text{ yr}^{-1} \text{ ha}^{-1}$ in the monocultures and $5 \text{ m}^3 \text{ yr}^{-1} \text{ ha}^{-1}$ in the mixed stands, distributed as Norway spruce $2.9 \text{ m}^3 \text{ yr}^{-1} \text{ ha}^{-1}$, oak $2 \text{ m}^3 \text{ yr}^{-1} \text{ ha}^{-1}$ and other species $0.1 \text{ m}^3 \text{ yr}^{-1} \text{ ha}^{-1}$.

12.7 Methods and models

The PROFILE model was used for calculating the weathering rates. The soils were sampled and the mineralogy calculated by the use of the Uppsala model. An ocular inspection by the team mineralogist was performed to detect the major minerals present. Already at sampling,

| Substance | Monoculture | Oak-spruce mixture | | |
|-----------|---------------------------|--------------------|--------|--------|
| | Norway spruce % weight | Oak | Spruce | Other |
| Ca | 0.11 | 0.140 | 0.12 | 0.12 |
| Mg | 0.012 | 0.01 | 0.01 | 0.022 |
| K | 0.062 | 0.1 | 0.048 | 0.11 |
| P | 0.0082 | 0.0073 | 0.0064 | 0.0124 |
| N | 0.062 | 0.14 | 0.054 | 0.12 |

Table 12.2: *The nutrient concentrations in stems at Jämjö, based on measurements on trees from the site (% weight). "Other" refers to the average employed for hornbeam, beech, birch, aspen and mountain ash.*

| Substance | Norway spruce | Oak | Substance | Norway spruce | Oak |
|-----------------|---------------------------------------|---|-----------|---------------------------------------|------|
| | kEq ha ⁻¹ yr ⁻¹ | | | kEq ha ⁻¹ yr ⁻¹ | |
| SO ₄ | 0.84 | 0.50 | Ca | 0.21 | 0.13 |
| NO ₃ | 0.81 | 0.49 | Mg | 0.12 | 0.07 |
| Cl | 0.48 | 0.28 | K | 0.067 | 0.04 |
| NH ₄ | 0.70 | 0.42 | Na | 0.39 | 0.21 |
| P | 0.03 | 0.02 | | | |
| Acidity | 2.07 | 1.30 | | | |
| | | m ³ m ⁻² yr ⁻¹ | | | |
| Precipitation | 0.44 | 0.62 | Runoff | 0.05 | 0.11 |

Table 12.3: *Deposition input data for Jämjö.*

feldspars, hornblende and epidote were spotted on the shovel. The soil texture was measured with a granulometric method. The net uptake was estimated from the nutrient contents and the different growth estimates made at the site.

12.8 Biogeochemical mapping of a single stand

Both Norway spruce and mixed species stands were biogeochemically mapped in terms of standing reservoirs above and below ground. Where direct measurement could not be carried out, combinations of data and values from the literature were used to obtain the best estimates available in terms present knowledge. The PROFILE model was used to calculate the weathering rate under different growth and deposition input conditions. Since the effective root depth of the stand was found to be a particularly important parameter in assessments of sustainability, an it warrants further consideration. It should be noted that the broadleaves cause a deeper total root penetration of the soil, although this may be considered a trivial fact (Ståål 1986; Almgren et al., 1984). It is also a cause of brown soil formation where deciduous tree cover is present. In brown soils which are planted with Norway spruce a process of podzolization

| Item | Spruce | Mixed | Unit |
|------------------|---------|--------|--------------------------|
| Stem mass | 148,500 | 80,000 | kg d.w. ha ⁻¹ |
| Stem mass Ca | 163 | 128 | kg d.w. ha ⁻¹ |
| Stem mass K | 92 | 83 | kg d.w. ha ⁻¹ |
| Stem mass Mg | 18 | 11 | kg d.w. ha ⁻¹ |
| Stem mass N | 92 | 95 | kg d.w. ha ⁻¹ |
| Stem mass P | 12 | 7 | kg d.w. ha ⁻¹ |
| Litterfall total | 4,410 | 4,360 | kg d.w. ha ⁻¹ |
| Litterfall Ca | 11.90 | 8.00 | g kg ⁻¹ |
| Litterfall Mg | 0.68 | 1.15 | g kg ⁻¹ |
| Litterfall K | 1.86 | 4.41 | g kg ⁻¹ |
| Litterfall N | 7.50 | 10.50 | g kg ⁻¹ |
| Litterfall P | 0.79 | 0.70 | g kg ⁻¹ |

Table 12.4: Amounts and fluxes in the stands at Jämjö.

immediately starts, taking 100 to 300 years to complete (Ugolini 1973). The effective root depth for the Norway spruce stands is estimated to be approximately 0.35 m and that for the mixed Oak-Norway spruce stands 0.75 m, see Fig. 12.5) and also the reasoning below. The root weight distributions obtained were consistent with data found in the literature (see, for example, the root distributions found in Höglwald in Bavaria, in Klosterhede in Denmark, and in Gårdsjön in Sweden, as given in the Forest Ecology and Management Journal's Special Issue on Nitrex-Exman in 1998; see also Carbonnier 1971, 1975; Kreutzer and Heil 1991, Bredemeier et al 1998, Skogshandboken 1985, Grönare skog 1999; Kimmins 1997). Nutrient uptake appear to approximately follow the fine root distribution, see Kreutzer and Heil (1991) and Warfvinge et al. (1998) for further discussion. Note that the total root mass per hectare is larger in the mixed stands, although the aboveground biomass there is lower than in the monocultures. This implies the root/shoot ratio to be larger in the mixed stands which means that there is a larger water and nutrient supply capacity there for sustaining a canopy. The following analyses are based on the data obtained and determining the effective soil depth for nutrient uptake by reading off the cumulative root mass value for each depth in question from Fig. 12.1:

| Tree species | Included root mass | | | |
|-------------------------------|-------------------------|------|------|------|
| | 80% | 85% | 90% | 95% |
| | Effective root depth, m | | | |
| Norway spruce monoculture | 0.22 | 0.24 | 0.3 | 0.39 |
| Oak-Norway spruce mixed stand | 0.37 | 0.40 | 0.48 | 0.61 |
| Oak stand | 0.40 | 0.47 | 0.54 | 0.87 |

Certain data in this field is lacking, i. e. on the relationship between root biomass at a given soil depth and nutrient uptake there, and no real regional surveys exist. Thus, the results must be treated as estimates made under conditions of uncertainty.

| Soil depth meter | Norway spruce | Spruce-Oak 4:6 mix kg ha ⁻¹ | Oak |
|---------------------|------------------|--|----------|
| 0-0.04 m | 450 | 250 | (110) |
| 0.04-0.14 m | 4,300 | 3,800 | (1,800) |
| 0.14-0.24 m | 1,450 | 4,000 | (7,030) |
| 0.24-0.34 m | 650 | 1,650 | (2,310) |
| 0.34-0.54 m | 450 | 1,650 | (2,450) |
| 0.54-0.84 m | (15) | (550) | (555) |
| 0.84-1.00 m | (0) | (150) | (150) |
| Sum | 7,315 | 12,050 | (14,405) |

Table 12.5: Measured total root distribution at Jämjö in kg root mass per hectare, based on samples taken from the Norway spruce stand and the mixed spruce-oak stand at Jämjö. As can be seen, the broad-leaved trees have a deeper total fine root penetration of the soil. Values in brackets are estimates.

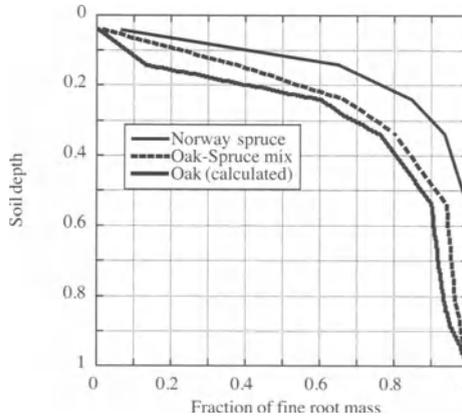


Figure 12.1: Cumulative root distribution for the Norway spruce monoculture and the Norway spruce-oak mixture. The values for oak were calculated by linear subtraction of the Norway spruce and renormalizing to 100%.

| Parameter | Unit | 1 | 2 | 3 | 4 |
|--------------------------|--------------------------------------|--------|-----------|-----------|-----------|
| Morphology | | O | E | E/B | B |
| Layer thickness | m | 0-0.04 | 0.04-0.14 | 0.14-0.24 | 0.24-0.34 |
| Moisture content | $\text{m}^3 \text{m}^{-3}$ | 0.3 | 0.25 | 0.2 | 0.2 |
| Soil bulk density | kg m^{-3} | 300 | 1200 | 1500 | 1510 |
| Mineral surface area | $\text{m}^2 \text{m}^{-3} * 10^{-6}$ | 0.52 | 1.06 | 1.3 | 1.3 |
| CO ₂ pressure | times ambient | 5 | 10 | 20 | 30 |
| DOC | mg l^{-1} | 13 | 5.3 | 2.9 | 1.3 |
| $\log K_{Gibbsite}$ | $\text{kmol}^2 \text{m}^{-3}$ | 6.5 | 7.6 | 8.6 | 9.2 |
| Inflow | % of precip | 100 | 50 | 25 | 15 |
| Percolation | % of precip | 50 | 25 | 15 | 12 |
| Mg+Ca+K uptake | % of total max | 20 | 30 | 30 | 20 |
| N uptake | % of total max | 30 | 60 | 10 | 0 |
| Soil mineralogy | | | | | |
| K-feldspar | % of total | 10 | 20 | 19 | 17 |
| Oligoclase | % of total | 17 | 34 | 34 | 32 |
| Hornblende | % of total | 0 | 0.62 | 0.87 | 1.6 |
| Epidote | % of total | 0 | 1.8 | 1.8 | 1.9 |
| Biotite | % of total | 0 | 0.9 | 1.2 | 1.1 |
| Muscovite | % of total | 0 | 0.75 | 0.4 | 0.2 |
| Vermiculite | % of total | 0 | 4.1 | 4.4 | 4.9 |
| Apatite | % of total | 0 | 0.74 | 0.7 | 0.7 |

Table 12.6: PROFILE input data for Norway spruce at Jämjö.

| Parameter | Unit | 1 | 2 | 3 | 4 | 5 |
|--------------------------|--------------------------------------|--------|-----------|-----------|-----------|-----------|
| Morphology | | O | E | E/B | B | B/C |
| Layer thickness | m | 0-0.04 | 0.04-0.14 | 0.14-0.24 | 0.24-0.34 | 0.34-0.74 |
| Moisture content | $\text{m}^3 \text{m}^{-3}$ | 0.3 | 0.25 | 0.25 | 0.2 | 0.2 |
| Soil bulk density | kg m^{-3} | 300 | 730 | 900 | 940 | 1480 |
| Mineral surface area | $\text{m}^2 \text{m}^{-3} * 10^{-6}$ | 5.99 | 1.02 | 1.27 | 1.31 | 1.48 |
| CO ₂ pressure | times ambient | 2 | 5 | 20 | 30 | 40 |
| DOC | mg l^{-1} | 24 | 8.3 | 5 | 3.7 | 2.1 |
| $\log K_{Gibbsite}$ | $\text{kmol}^2 \text{m}^{-3}$ | 6.5 | 7.5 | 8.5 | 9.2 | 9.5 |
| Inflow | % of precip | 100 | 50 | 25 | 20 | 16 |
| Percolation | % of precip | 50 | 25 | 20 | 16 | 16 |
| Mg+Ca+K uptake | % of total max | 20 | 20 | 20 | 20 | 20 |
| N uptake | % of total max | 30 | 35 | 25 | 5 | 5 |
| Soil mineralogy | | | | | | |
| K-feldspar | % of total | 9.6 | 19 | 20.4 | 18.5 | 18.4 |
| Oligoclase | % of total | 15.3 | 31.5 | 35.6 | 31.6 | 36.1 |
| Hornblende | % of total | 0 | 1.31 | 0.89 | 1.97 | 1.15 |
| Epidote | % of total | 0 | 1.53 | 1.77 | 1.81 | 1.9 |
| Biotite | % of total | 0 | 0.37 | 0.79 | 0.53 | 1.11 |
| Muscovite | % of total | 0 | 0.86 | 0.66 | 0.59 | 0.17 |
| Vermiculite | % of total | 0 | 2.6 | 3.34 | 4.11 | 3.48 |
| Apatite | % of total | 0 | 0.36 | 0.33 | 0.41 | 0.38 |

Table 12.7: PROFILE input data for the mixed species stands at Jämjö.

| | pH | LOI % | CEC | Al | Ca mEq kg ⁻¹ | Mg | K | BS % | C g kg ⁻¹ | N g kg ⁻¹ |
|---------------------------------|------|----------|-------|-----|----------------------------|-------|-------|---------|-------------------------|-------------------------|
| Norway spruce monoculture stand | | | | | | | | | | |
| Humus | 4.43 | 27.9 | 124.0 | 363 | 1126 | 102.0 | 193.0 | 58.0 | 100.0 | 5.3 |
| 0-0.1 | 4.61 | 8.5 | 38.2 | 223 | 157 | 12.6 | 22.6 | 27.0 | 45.0 | 2.1 |
| 0.1-0.2 | 4.66 | 5.9 | 19.6 | 118 | 87 | 4.9 | 6.1 | 27.7 | 29.5 | 1.8 |
| 0.2-0.3 | 5.06 | 4.6 | 15.9 | 81 | 100 | 4.2 | 5.8 | 36.4 | 22.0 | 1.3 |
| 0.3-0.5 | 4.92 | 4.0 | 11.2 | 67 | 53 | 2.7 | 3.1 | 28.3 | 16.0 | 1.0 |
| Norway spruce-oak mixed stand | | | | | | | | | | |
| Humus | 4.20 | 47.0 | 162.0 | 503 | 1283 | 225.0 | 316.0 | 54.8 | 210.0 | 9.7 |
| 0-0.1 | 4.47 | 13.0 | 54.7 | 313 | 206 | 41.0 | 57.3 | 28.6 | 66.0 | 3.0 |
| 0.1-0.2 | 4.53 | 8.8 | 31.7 | 200 | 100 | 18.7 | 30.3 | 24.5 | 40.0 | 1.8 |
| 0.2-0.3 | 4.91 | 6.5 | 23.4 | 142 | 78 | 15.5 | 26.5 | 25.3 | 34.0 | 1.6 |
| 0.3-0.5 | 4.82 | 5.0 | 17.5 | 109 | 60 | 10.8 | 14.5 | 24.6 | 24.0 | 1.1 |

Table 12.8: Soil data from the Jämjö experiment, separated into horizons down to 0.5 m soil depth, from the Norway spruce stands (top) and the mixed stands (bottom). LOI is loss on ignition, BS is base saturation. All adsorbed elements were determined in BaCl₂ extractions.

| Depth meter | Ca | Mg keq ha ⁻¹ yr ⁻¹ | K keq ha ⁻¹ yr ⁻¹ | P | Ca | Mg keq ha ⁻¹ yr ⁻¹ | K keq ha ⁻¹ yr ⁻¹ | P |
|----------------|-------------------------------|---|--|-------|---------------------|---|--|-------|
| | Mixed Norway spruce-Oak stand | | | | Norway spruce stand | | | |
| 0-0.04 | 0.002 | 0.000 | 0.002 | 0.000 | 0.001 | 0.000 | 0.001 | 0.000 |
| 0.04-0.14 | 0.025 | 0.006 | 0.019 | 0.004 | 0.046 | 0.010 | 0.020 | 0.020 |
| 0.14-0.24 | 0.037 | 0.010 | 0.025 | 0.005 | 0.066 | 0.015 | 0.022 | 0.031 |
| 0.24-0.34 | 0.036 | 0.013 | 0.017 | 0.009 | 0.057 | 0.015 | 0.017 | 0.026 |
| 0.34-0.74 | 0.222 | 0.070 | 0.120 | 0.035 | | | | |
| Sum | 0.322 | 0.100 | 0.182 | 0.053 | 0.170 | 0.039 | 0.060 | 0.078 |

Table 12.9: The weathering rate in keq ha⁻¹yr⁻¹ for the Norway spruce stands and the Norway spruce-Oak mixed stands as calculated with PROFILE.

12.9 Assessment of nutrient sustainability

The weathering rate was calculated, with the PROFILE model, as the weathering per cation, chemical reaction, soil mineral, and soil layer. According to our estimates the weathering rate in Jämjö is caused to 22% by reactions with the acidity of the soil, to 33% by hydrolysis with water, to 33% by reactions with carbon dioxide, and to 10% by reactions with the organic acids in the soil. The major factors affecting the weathering rate can be said to be the rich mineralogy, the soil moisture and soil texture. The results are shown in Tab. 12.9. As can be seen, the weathering rate increases with depth down to a depth of 0.6 m in the soil, mainly because of the increase in mineral content. The weathering is of the same order of magnitude as the current uptake. The consistency of the calculations was checked against sodium concentrations. For the spruce stands, the correlation was excellent, whereas the measured sodium flux in the mixed stands was lower than the input would imply. Chloride was used to calibrate the water balance, and sulphate as an independent control of that calibration. Sustainability was assessed by calculating the mass balances for each of the nutrients at Jämjö. The results are shown in Tabs. 12.10 to 12.12. Three different scenarios were considered;

- The estimated rotation average growth rate together with the current weathering rates. Information contained in Mr. Ståål's personal notes was a help in making the estimates.
- The current growth rate which is approximately 40% larger than the historical rate.
- A hypothetical scenario in which the Norway spruce stand was allowed to occupy the soil of the mixed stand, and the mixed stand was allowed to occupy the soil of the Norway spruce stand. This was done to eliminate in so far as possible the differences in basic conditions between the sites.

The current leaching rate was used for all cases. The sustainable yield is defined as the maximum yield that can be obtained on the basis of the nutrients available in the system. In estimating this, sustainability is measured as long term sustainable yield Y , with respect to Ca, Mg, K and P. The equation employed was:

$$Y_{BC} = \min_i \left(\frac{BC_{U,i}^{crit} \cdot E_i}{\rho \cdot x_i} \right) \quad (12.8)$$

where $BC_{U,i}^{crit}$ is the critical uptake of element i =Ca, Mg, or K; E_i is the equivalent weight of element i ; ρ is the specific density of the harvested biomass; and x_i is the concentration of element i in the harvested biomass. The stand averages for both specific density and stem nutrient concentrations in the mixed stands differ, depending on the relative amounts of the different species. Hence, in the rotation average growth rate scenario both the average wood density and the average stem nutrient concentrations are lower than in current growth rate scenario since the share of Norway spruce in the mixed stands is greater in rotation average growth rate scenario than in for growth in the current situation. The critical uptake is limited by the nutrients available;

$$BC_{U,i}^{crit} = W_i + D_i - L_{min} \quad (12.9)$$

where W_i is the weathering release of element i , D_i is the atmospheric deposition of element i , and L_{min} is the minimum rate of leaching from the system. The units in the tables are in kEq $\text{ha}^{-1}\text{yr}^{-1}$ and the yields are expressed as $\text{m}^3 \text{ha}^{-1}\text{yr}^{-1}$. According to our calculations, assuming no nutrients are added, the sustainable yield in the Norway spruce stands is $5.4 \text{ m}^3 \text{ha}^{-1}\text{yr}^{-1}$ and that of the mixed stands is $6.9 \text{ m}^3 \text{ha}^{-1}\text{yr}^{-1}$.

Although, at the current growth rates, both stand types show negative nutrient balances, the deficit is greater in the monocultures than in the mixed stands. As can be seen in Tables (12.10 to 12.12) the mineralogy is richer and the soil bulk density is greater in the monocultures than in the mixed-species stands. This is in agreement with observations made in the field. If

| Flux | Norway spruce | | | | Oak-Norway spruce | | | |
|----------------------|---------------------------------------|-------|-------|--------|-------------------|-------|-------|-------|
| | Ca | Mg | K | P | Ca | Mg | K | P |
| | keq ha ⁻¹ yr ⁻¹ | | | | | | | |
| Weathering rate | 0.173 | 0.039 | 0.060 | 0.026 | 0.306 | 0.095 | 0.180 | 0.048 |
| Deposition | 0.210 | 0.115 | 0.067 | 0.030 | 0.130 | 0.069 | 0.040 | 0.020 |
| Leaching with runoff | 0.265 | 0.090 | 0.007 | 0.004 | 0.200 | 0.070 | 0.126 | 0.004 |
| Future harvest | 0.220 | 0.039 | 0.063 | 0.053 | 0.172 | 0.023 | 0.049 | 0.030 |
| Balance | -0.102 | 0.025 | 0.057 | -0.001 | 0.064 | 0.071 | 0.045 | 0.034 |
| Sustainable yield | 5.373 | 16.32 | 18.95 | 9.779 | 6.880 | 20.56 | 9.565 | 10.71 |
| Actual growth | 10 | | | | 5 | | | |

Table 12.10: *ROTATION AVERAGE GROWTH SCENARIO; Calculation of mass balances at Jämjö using the rotation average growth rate estimated from Stål (1986) and by the authors. Units are in keq ha⁻¹yr⁻¹ and yields are expressed as m³ ha⁻¹yr⁻¹. At present, the forest is growing faster than in the past. The sustainable yield calculated assumes no nutrient additions to have been made.*

the mineralogy, the surface area, and the bulk density in the spruce monocultures and in the mixed stands are switched, the sustainable yield for the spruce monocultures decreases to 1.8 m³ha⁻¹yr⁻¹, whereas the sustainable yield for the mixed-species stands increases to 6.5 m³ ha⁻¹yr⁻¹. Hence, if the soil conditions are the same, the sustainable yield is approximately 2.5 m³ ha⁻¹yr⁻¹ to 3 m³ ha⁻¹yr⁻¹ higher in the mixed species stands than in the spruce monocultures. However, if the objective is to assess the nutrient sustainability of the management system employed the rotation average growth rate scenario is more accurate. This means a sustainable rotation average growth rate of approximately 6.9 m³ha⁻¹yr⁻¹ in the mixed stands and 5.4 m³ ha⁻¹yr⁻¹ in the spruce monocultures (Table 12.10). This suggests the mixed stands to be sustainably managed, but the monocultures to not be. The calculations also suggest that in the mixed-species stands production can be increased by almost 2 m³ ha⁻¹yr⁻¹ without depleting soil nutrient resources. The model calculations reproduced the concentrations of Ca, Mg, K, SO₄, Cl and Na rather accurately, suggesting the weathering calculations in PROFILE to be robust for this type of soil. The Cl concentrations were used to calculate the soil hydrology input. The model suggests that the site is only slightly affected by acidification, and that the aluminium concentrations are well below what is considered harmful to trees. It should be observed that the nitrate leaching as modeled agreed well with the lysimeter data. For the mixed stands in contrast to the monocultures, the contribution of nutrients from weathering increased by between 70% and 200%, the nutrient input through deposition decreased by approximately 40%, the leaching losses decreased by 25% in the case of Ca and Mg, but increased by several hundred percent in the case of K, and the nutrients lost at future harvests decreased by between 20% and 40%. The high leakage of K from the mixed stands could be caused by the high mobility of the K in the soil and a low demand for K. Root bioassays showed the demand for K to be 40% higher in the fine root material from the monocultures than in the fine root material from the mixed stands (Sudhaus 1998). In addition, K concentrations in current year

| Flux | Norway spruce | | | | Oak-Norway spruce | | | |
|----------------------|---------------------------------------|-------|-------|--------|-------------------|-------|--------|-------|
| | Ca | Mg | K | P | Ca | Mg | K | P |
| | keq ha ⁻¹ yr ⁻¹ | | | | | | | |
| Weathering rate | 0.173 | 0.039 | 0.060 | 0.026 | 0.306 | 0.095 | 0.180 | 0.048 |
| Deposition | 0.210 | 0.115 | 0.067 | 0.030 | 0.130 | 0.069 | 0.040 | 0.020 |
| Leaching with runoff | 0.265 | 0.090 | 0.007 | 0.004 | 0.200 | 0.070 | 0.126 | 0.004 |
| Future harvest | 0.329 | 0.059 | 0.095 | 0.080 | 0.325 | 0.043 | 0.109 | 0.057 |
| Balance | -0.211 | 0.005 | 0.025 | -0.028 | -0.089 | 0.051 | -0.015 | 0.007 |
| Sustainable yield | 5.373 | 16.32 | 18.95 | 9.779 | 5.671 | 17.20 | 6.733 | 8.773 |
| Actual growth yield | 15 | | | | 7.8 | | | |

Table 12.11: *CURRENT SCENARIO; Calculation of mass balances at Jämjö using the standard yield and the present standing biomass. Units are in keq ha⁻¹yr⁻¹ and yields are expressed as m³ ha⁻¹yr⁻¹.*

needles were higher in the spruces in the mixed stands than in the monocultures. The leaching of K, in contrast to the leaching of Al or N, for example, is not a problem as long as the inputs of K to the system balance the outputs, as is the case in the mixed stands in Jämjö (Tabs. 12.10-12.12). From the figures above it appears that the difference in the weathering rate is the most potent factor in assessing the nutrient sustainability. In fact, the weathering rate itself is not the decisive factor, but rather whether or not the weathered material is available for uptake. This, in turn, is strongly influenced by the presence or absence of roots and by their uptake ability. In the present study we implicitly assumed there to be a close correlation between root presence, as estimated from field data (effective rooting depth, Figure 12.1), and the availability and uptake of weathered material. Since this may not always be the case, however, there is a definite need for research on this matter. According to the mass balance calculations, if the trees in the monocultures do have access to nutrients below what appeared to be the effective rooting depth at the site, sustainable yield in the monocultures could be increased. For example, if the spruces in the monocultures could access 50% of the weathered material below the 0.34 m level in the mixed stands (Table 12.10-12.12), a rotation average growth rate of close to 10 m³ ha⁻¹yr⁻¹ would be sustainable. According to the present knowledge however, this is not likely to be the case. Rooting depth depends not only on species, but also on soil depth and moisture. Such factors often play a more important role in determining the rooting depth than the species does. On waterlogged soils, for example, the superficial water table restricts root growth in deep soil layers, regardless of species. It should be emphasized that higher growth rates lead to a greater removal of nutrients per time unit, as was the case in the monocultures at Jämjö. Having mixed-species stands, in which production of quality is emphasized, is more likely for this reason to be a biogeochemically sustainable form of management than that of having monocultures that are managed with the primary aim of achieving quantity rather than quality. The choice of harvesting method also has a strong influence on the mass balances, nutrient removal increasing markedly when whole tree harvesting takes place. In addition, leaching losses can be kept to a minimum by shelter-woods being left instead of using clear-cutting. The assessment of nutrient sustainability involves only the base cations and P. However, the forest management

| Flux | Norway spruce | | | | Oak-Norway spruce | | | |
|----------------------|---------------------------------------|--------|-------|--------|-------------------|-------|--------|-------|
| | Ca | Mg | K | P | Ca | Mg | K | P |
| | keq ha ⁻¹ yr ⁻¹ | | | | | | | |
| Weathering rate | 0.094 | 0.024 | 0.040 | 0.030 | 0.341 | 0.112 | 0.191 | 0.079 |
| Deposition | 0.210 | 0.115 | 0.067 | 0.030 | 0.130 | 0.069 | 0.040 | 0.020 |
| Leaching with runoff | 0.265 | 0.090 | 0.007 | 0.004 | 0.200 | 0.070 | 0.126 | 0.004 |
| Future harvest | 0.329 | 0.059 | 0.095 | 0.080 | 0.325 | 0.043 | 0.109 | 0.057 |
| Balance | -0.290 | -0.010 | 0.005 | -0.024 | -0.054 | 0.068 | -0.004 | 0.038 |
| Sustainable yield | 1.776 | 12.50 | 15.79 | 10.53 | 6.512 | 20.31 | 7.521 | 13.02 |
| Actual growth yield | 15 | | | | 7.8 | | | |

Table 12.12: *CHANGE SOIL SCENARIO*; Calculation of mass balances at Jämjö for the hypothetical case in which the locations of the Norway spruce and the mixed species stands are exchanged. Units are in keq ha⁻¹yr⁻¹, the yields are expressed as m³ ha⁻¹yr⁻¹. The table shows that the difference between the mixed stands and the Norway spruce stands is systematic and persists, even when the two stands exchange soils.

may be adjudged to be sustainable in terms of there being no negative balance for Ca, K, Mg, or P, but to not be sustainable in terms of micro-nutrients. The risks of micro-nutrient deficiencies has not been investigated very much in southern Swedish forests. However, since several cases of B and Cu deficiencies in Norway spruce and in Scots pine have been found there it is possible that these deficiencies are more common in southern Sweden than assumed so far (Thelin 2000). If so, one could reasonably expect there to be greater risk of negative balances of B and Cu in Norway spruce monocultures than in mixed stands due to the removal of nutrient capital at harvest being greater and the effective rooting depth being lower.

12.10 Assessment of economic sustainability

Assessment of the economic sustainability of different management scenarios at Jämjö needs to take into account the degree to which these stay within the sustainability limits of the site. The management scenarios that are compared are Norway spruce monocultures (Table 12.15) and mixed oak/spruce stands (Table 12.16), with and without natural regeneration. As can be seen in table 12.15, the spruce stands are managed in the conventional way by there being four operations between regeneration and harvest and bulk volume production being emphasized. The mixed stands are much more labour intensive, ten operations between regeneration and harvest being required. The focus there is on the high quality oakwood contained in the 40 to 50 primary stems per hectare. These stems are managed on almost an individual basis involving the clearing and thinning of competing stems, as well as stems sheltering and pruning. The effective interest rates as well as changes in the market price of timber over the years pose considerable problems when assessing economic sustainability of different management scenarios. Assessments quickly become a comparison of tables of exponentially escalating interest rates, detracting attention from what one really wants to compare. More importantly, the small forest owners apparently do not assess economic matters in this way. Interviews the authors conducted

| Trees species | Rotation time years | Standard Yield $\text{m}^3\text{ha}^{-1}\text{yr}^{-1}$ | Sustainable Yield $\text{m}^3\text{ha}^{-1}\text{yr}^{-1}$ | Observed growth $\text{m}^3\text{ha}^{-1}\text{yr}^{-1}$ |
|-------------------------|---------------------------|---|--|--|
| Norway spruce | 70 | 10 | 5.4 | 15 |
| Oak-Spruce mixed | 120 | 5.5 | 6.9 | 7.8 |
| - from Norway spruce | 70 | 2.9 | | 5.3 |
| - from Oak | 120 | 2.0 | | 2.0 |
| - from Hornbean and Ash | 100 | 0.1 | | 0.1 |

Table 12.13: *Comparison of the basic properties of the stands. Standard yield corresponds to what in Swedish is termed "bonitet"; it represents the maximum rate of production of a stand. Sustainable yield is based on the mass balances for Ca, Mg, K and P.*

with more than 150 forest owners in the region around Emmaboda and between Bromölla and Tingsryd during a lecture series they held there in 2000 and 2001, suggest that conservation of capital and the building of hidden reserves before generation shifts occur at the farms to be the driving aims behind forest management decisions in such cases. This is a far cry from the theoretical corporate economics usually applied in this area. Corporate philosophy is almost invalid within such systems. For these reasons a radical approach to the problem was taken. We avoided the interest rate problem by converting all measures of cost and of income into Norway spruce pulpwood equivalents (m^3 SPE), and of assuming the ratios between the prices to remain constant during the 120 years. We also implicitly assume that a cubic meter of wood is of equal interest economically regardless of what year it can be harvested, and that computations concerning one type of wood can be converted into those concerning another by using constant price ratios. Some of the assumptions concerning price structures that were made are listed in Table 12.14. Although, in the deciduous scenarios, it is generally assumed that fencing is necessary in order to prevent excessive browsing, at Jämjö the land is rich in deciduous tree juveniles making it questionable whether fencing is justified. A significant difference between the Norway spruce stands and the spruce-oak stands is the longer rotation time in the mixed stands. We set the rotation time for a Norway spruce stand to be 70 years, whereas for the mixed stands we assume the rotation time to be 120 years.

The results for the Norway spruce are shown in Table 12.15. The net productivity during the period was $10 \text{ m}^3\text{ha}^{-1}\text{yr}^{-1}$. The sustainable yield is $5.4 \text{ m}^3\text{ha}^{-1}\text{yr}^{-1}$; the uptake in excess of that, corresponding to $4.6 \text{ m}^3\text{ha}^{-1}\text{yr}^{-1}$, must be replaced at the end of the period. The assessments for the mixed stand are shown in Table 12.16. The net productivity during the period was 360 m^3 or $3.0 \text{ m}^3\text{ha}^{-1}\text{yr}^{-1}$ for the Norway spruce and 240 m^3 or $2.0 \text{ m}^3\text{ha}^{-1}\text{yr}^{-1}$ for the oak. No excess nutrients have been removed. The profit yield during the 120-year period, expressed in Norway spruce pulpwood equivalents, is $2,215 \text{ SEK ha}^{-1}\text{yr}^{-1}$.

With the inclusion of spruce in the mixed stands, there is a positive economic result already after 60 years, comparable with that of the spruce monoculture (Table 12.16). If the oak had been managed as monocultures the economic result would not have been positive until the time of harvest in year 120, i. e. 60 years later than for the mixed alternative; the capital requirement is increased, the profit is reduced to $1,756 \text{ kr ha}^{-1}\text{yr}^{-1}$, and the costs of pruning are higher in an oak monoculture since there is no stem-sheltering undergrowth. Thus, the yearly profit for the oak monoculture alternative would probably be more than 500 SEK less than for the mixed stands. Clearly, the costs involved in managing a mixed stand are more than balanced out by the income obtained from the thinnings of the spruce, the overall costs thus

| Trees species | Fuel wood | Pulp wood | Timber | Fuel wood | Pulp wood | Timber |
|---------------|------------------------------|-----------|---------|-----------------------|-----------|--------|
| | Price in SEK m ⁻³ | | | Relative price to SPE | | |
| Norway spruce | 100.- | 250.- | 400.- | 0.4 | 1.0 | 1.6 |
| Scots pine | 100.- | 200.- | 500.- | 0.4 | 0.8 | 2.0 |
| Birch | 150.- | 250.- | 600.- | 0.6 | 1.0 | 1.6 |
| Beech veneer | 150.- | 280.- | 1,500.- | 0.6 | 1.12 | 6.0 |
| Beech A | 150.- | 280.- | 1,000.- | 0.6 | 1.12 | 4.0 |
| Beech B | 150.- | 280.- | 500.- | 0.6 | 1.12 | 2.0 |
| Oak veneer | 150.- | -- | 3,250.- | 0.6 | -- | 13.0 |
| Oak A | 150.- | -- | 1,500.- | 0.6 | -- | 6.0 |
| Oak B | 150.- | -- | 900.- | 0.6 | -- | 4.0 |
| Oak C | 150.- | -- | 500.- | 0.6 | -- | 2.0 |

Table 12.14: *To the left is shown the price structure used for the economic assessment made in Swedish kronor (SEK) per cubic meter. The prices were estimated from market prices for the spring of 2000. The prices are approximate. To the right, the relative price structure per cubic meter is shown, normalized to the Norway spruce pulpwood price (SPE), as used for the economic assessments. It is assumed that these ratios remain constant during the period of 2000 to 2120.*

being much lower than for the oak monoculture alternative. If natural regeneration, rather than plantation, is possible (soil scarification is still considered necessary) the yearly profits increase for all the alternatives (Table 12.16 and 12.17). However, the differences in yearly profits between the different management scenarios are rather small in this respect. The differences become greater if the economic results are calculated in a more traditional way, e. g. 3% yearly discount of costs and incomes. In that case the mixed stands are only competitive if natural regeneration is possible, rather than planting and fencing being required. Thus, for large forest owners who adhere to corporate philosophy in assessing their economic results spruce would be considered superior if planting was required. Studies of similar mixed oak-spruce stands close to Jämjö indicate that spruce production can be increased to up to 600 m³ per hectare and rotation without oak development being impeded (Linden et al., 2001). Thus, the mixed stand alternative appears to be superior, regardless of how the economic result is calculated. In the present comparisons, the differences in capital requirements are much greater than the differences in yearly profits. Although the plantation costs in all the alternatives are comparable, the fencing necessary in the mixed stand and the oak alternative increases the regeneration costs considerably. The capital requirements and how long it takes until profit can be made are probably more important than yearly profit for many small-scale forest owners in assessing the economic consequences of different forest management alternatives. Thus, if the starting point is a clear-cut area and both planting and fencing are required, most small-scale forest owners would be drawn by short-term economic considerations to planting spruce - the investment threshold for the establishment of oak stands being too high. Therefore, if the Swedish government wants to influence forest managers in southern Sweden to plant more deciduous and mixed forests subsidies for the planting of deciduous seedlings and for fencing are required. The calculations above do not consider the costs for maintaining a high level of

| Time | Action | Yield m ⁻³ | Timber % | Cost | In | Net |
|------|----------------------|--------------------------|-------------|------|---------------------|-----|
| | | | | | m ⁻³ SPE | |
| 0 | Soil scarification | | | 6 | | -6 |
| 0 | Planting of spruce | | | 40 | | -46 |
| 10 | Clearing | | | 12 | | -58 |
| 30 | Thinning | 40 | 0 | 32 | 40 | -50 |
| 40 | Thinning | 70 | 10 | 36 | 74 | -12 |
| 55 | Thinning | 90 | 20 | 47 | 101 | 42 |
| 70 | Harvest | 500 | 65 | 140 | 695 | 597 |
| 70 | Nutrient replacement | | | 14 | | 583 |
| 70 | Total | 700 | | 327 | 910 | 583 |

Table 12.15: *NORWAY SPRUCE; Profit and balance estimates for the Norway spruce alternative at Jämjö. The net productivity during the period was 10 m³ha⁻¹yr⁻¹. The sustainable yield is 5.4 m³ha⁻¹yr⁻¹, uptake in excess of this being replaced. The profit yield over 120 years in terms of Norway spruce pulpwood equivalents is 10 m³ ha⁻¹yr⁻¹ or 2,082 SEK ha⁻¹yr⁻¹. Assessments are made for one hectare, the sequence being started by an empty clear-cutting.*

biodiversity. It is probable that these are higher in a Norway spruce monoculture alternative. Many insects and lichens are associated with old oak stems. To ensure their possibilities of remaining in the area it is necessary to let a few stems per hectare remain as "eternal" trees. However, it is probable that fewer stems would need to be set aside in this way in the oak/spruce mixed alternative than in the spruce monocultures since younger oak stems could serve to relay insects and lichens to older stems further away. The approximate costs of biodiversity in Jämjö could be estimated to be as follows:

- For the mixed stands leaving 3 oaks per hectare as old biodiversity bearers would mean a profit reduction of 3 out of an average of 44 oaks per ha, and would involve a slight reduction in the costs for management (3 SEK ha⁻¹yr⁻¹). Together this represents a profit reduction of 6.6% of the oak share or 123 SEK ha⁻¹yr⁻¹.
- For the monocultures an area corresponding to 5 oaks per hectare would be removed from commercial use and be left for producing biodiversity. This would represent a profit reduction of 11.1% or 238 SEK ha⁻¹yr⁻¹.

The use of mixed-species stands also means a spreading of risks as compared with the use monocultures. Butt rot, which is considered one of the most serious problems for spruce in southern Sweden, can reduce the amount of harvestable wood by as much as 20%. This would mean a reduction in profits by 20% in the monocultures but only a 3% reduction in the mixed stands, since only spruce is affected and since most of the income is obtained from oak wood.

12.11 Conclusions

We can conclude that both the Norway spruce stands and the mixed stands in Jämjö have a sustainable harvest rate of approximately 5.5 m³ha⁻¹yr⁻¹. Since the Norway spruce stands at present are growing at a rate of 10 m³ha⁻¹yr⁻¹ a corresponding amount of nutrients needs to be added. Otherwise the present rate of growth would exhaust or deplete the soil when harvesting takes place. A hypothetical switching of tree species to opposite stand locations,

| Time | Action | Spruce m ⁻³ | Timber % | Oak m ⁻³ | Veneer % | A % | B % | C % | FW % | Cost | In m ⁻³ SPE | Net |
|------|--------------------|---------------------------|-------------|------------------------|-------------|--------|--------|--------|---------|------|---------------------------|-------|
| 0 | Soil scarification | | | | | | | | | 6 | | -6 |
| 0 | Planting | | | | | | | | | 48 | | -54 |
| 0 | Fencing | | | | | | | | | 40 | | -94 |
| 5 | Clearing | | | | | | | | | 12 | | -106 |
| 15 | Clearing | | | | | | | | | 12 | | -118 |
| 20 | Pruning | | | | | | | | | 4 | | -122 |
| 30 | Thinning | 50 | 0 | 0 | | | | | | 40 | 50 | -112 |
| 40 | Thinning | 50 | 10 | 10 | 0 | 0 | 0 | 10 | 90 | 35 | 60 | -87 |
| 40 | Pruning | | | | | | | | | 4 | | -91 |
| 50 | Thinning | 60 | 20 | 10 | 0 | 0 | 20 | 30 | 50 | 40 | 83 | -47 |
| 60 | Thinning | 70 | 30 | 20 | 0 | 10 | 30 | 30 | 30 | 48 | 132 | 36 |
| 70 | Thinning | 130 | 50 | 30 | 0 | 20 | 40 | 20 | 20 | 86 | 264 | 214 |
| 80 | Pruning | | | | | | | | | 40 | | 174 |
| 120 | Harvest | 0 | 0 | 170 | 25 | 25 | 10 | 20 | 20 | 68 | 957 | 1063 |
| 120 | Total | 360 | | 170 | | | | | | 483 | 1546 | 1,063 |

Table 12.16: *OAK-NORWAY SPRUCE MIXED STAND; Profit and balance estimates for the mixed forest alternative at Jämjö. The net productivity during the period was 360 m³ or 3 m³ha⁻¹yr⁻¹ for Norway spruce and 170 m³ or 2 m³ha⁻¹yr⁻¹ for oak. No excess nutrients were removed. The profit yield for a 120-year period in terms of Norway spruce pulpwood equivalents is 9 m³SPE ha⁻¹yr⁻¹ or 2,215 SEK ha⁻¹yr⁻¹. Assessments are made for one hectare, the sequence being started by an empty clear-cutting.*

suggested a mixed Norway spruce-oak stand to have a sustainable yield approximately 1-1.5 m³ha⁻¹yr⁻¹ greater than that of the Norway spruce monoculture under the same conditions. A final comparison of economic performance is shown in Table 12.17. As can be seen, the Norway spruce stands and the mixed stands are comparable in performance, that of the mixed stand perhaps being somewhat better. Natural regeneration does have a strong impact on the profitability of the stands. Although natural regeneration may prolong the rotation time, this is often not the case. We conclude that the frequently claimed economic superiority of Norway spruce in terms of forest productivity appears to not be supported here. The superiority is only very short-term, being achieved through overexploitation of the site resources and depletion of the soil. The value produced by deciduous trees is economically equivalent or even better when the sustainability potential is used as a limiting condition in the system. In the present studies we have assumed the roots to be 100% successful in nutrient uptake, although studies at the Asa Forest Research Park imply this to be too optimistic. The results there indicates that Norway spruce has a root uptake efficiency of 85-90% and deciduous trees an uptake efficiency of 95-97%. We attribute this difference to roots not being present in the soil volume as a whole at all times, the higher efficiency of deciduous trees being caused by their deeper rooting. This suggests the long-term economic benefits of forestry involving deeply rooted tree species and mixed-species stands. The Jämjö study shows that a rotation average growth rate of 10 m³ha⁻¹yr⁻¹ may not be sustainable in Norway spruce monocultures despite that the relatively rich mineralogy would imply so. Weathering and deposition would be unable to compensate for losses of Ca in leaching and in stem uptake. However, sustainable production appears to be greater in mixed-species stands dominated by oak and spruce than in Norway spruce monocultures. The most important factor here appears to be the greater rooting depth in the mixed-species stands and the access thus provided to pools of nutrients from weathering found at soil depths below the rooting zone of Norway spruce. In an environment with a high deposition of N and S, such as in southern Sweden, soil acidification and increased growth due to nitrogen deposition

| Trees species | Yearly profit SEK ha ⁻¹ | Capital requirement SEK ha ⁻¹ | Time to positive balance, yr |
|----------------------|---------------------------------------|---|---------------------------------|
| Norway spruce | | | |
| Planted | 2,082.- | 14,500.- | 55 |
| Natural regeneration | 2,225.- | 4,500.- | 40 |
| Oak-Spruce mix | | | |
| Planted and fenced | 2,215.- | 30,500.- | 60 |
| Natural regeneration | 2,398.- | 8,500.- | 50 |
| Oak-Beech | | | |
| Planted and fenced | 1,756.- | 32,000.- | 120 |
| Natural regeneration | 1,936.- | 10,000.- | 70 |

Table 12.17: *Comparisons of the different alternatives tested. The hardwood forest binds more capital for a substantially longer period of time, but provides a substantially better yield in terms of Norway spruce pulpwood equivalents for the period as a whole.*

and intensified forest management removes much of the nutrient capital in the superficial soil layers. Hence, the sustainable production of shallow-rooted species there may be problematical. The present investigation of the economic performance of mixed-species stands and of Norway spruce monocultures in Jämjö showed the yearly profit to be slightly higher in the mixed stands assuming that one m³ of wood is of equal interest regardless of when it is harvested. However, the capital requirements for the mixed stands are greater, especially if planting and fencing is required. The inclusion of Norway spruce in the oak stands serves as an economic buffer early in the rotations. Although the lengths of the time until the income exceeds the cost are comparable in the case of Norway spruce monocultures and of mixed-species stands, they are much longer in oak monocultures. Thus, oak/spruce mixed stands may be a more sustainable alternative than Norway spruce monocultures in terms both of economic performance and of the preservation of long-term soil fertility.

Chapter 13

Productivity scenarios for the Asa Forest Park

Assessment of sustainability aspects for production scenarios

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13.1 Asa site description

The primary focus of the SUFOR research programme was the Asa Royal Forest Park, 33 km² in size, in which the Asa Forest Research Park 12 km² in size, is included. A field station of the Swedish Agricultural University is located in the park, which was expropriated from its German owner in 1945 and become a Royal domain. The area is host to a number of research projects and conditions for synergistic cooperation is good. Technicians from the station assisted in sampling and data collection. Various parameters of the area are shown in Tab. 14.2. The park borders to Lake Asa, the majority of the area being found on the western side of the lake. A map of the area is shown in Fig. 13.1. The park is located 40 km north of the city of Växjö in the province of Småland. Forestry in the area is traditional conifer forestry, representing the point of departure for any future changes. In the past, the original forest cover in Asa was a mixed deciduous forest dominated by lime (*Tilia*) and alder (*Alnus*), and also containing European beech (*Fagus sylvatica*), Scots pine (*Pinus sylvestris*), oak (*Quercus robur*) and birch (*Betula*). Farming on the shores of the lake started around 1200 AD and Norway spruce entered the landscape on a small scale around 1600. Mixed deciduous vegetation dominated until clearance of the land by farmers from about 1700 onwards. After 1840, Norway spruce increased markedly through colonization and through planting on land that had been abandoned after large numbers of emigrants went to America. Small crofts and farms were reforested, a process that in the province of Småland continued as a result of urbanization from about 1930 on until the present day. From now on conifers, and Norway spruce in particular, are dominant. Thus, the ecosystem at Asa has undergone very large changes, and can be said to be completely a product of cultural activities during the last 700 years. The Asa forest estate is 3,288 ha in size and is owned by the Swedish State (Sveaskog AB). Part of the forest is used presently as an experimental forest by the Swedish University of Agricultural Sciences (1,080 ha), and by the National Board of Forestry for educational purposes (393 ha). The remaining 1,815 ha being used for commercial forestry. Meteorological conditions have been recorded at the Asa Forest Research Station since 1988. The average annual temperature during the period of 1988-1999 was 6.4 °C, the coldest and warmest months (January/July) having had average temperatures of -1 °C and +16 °C, respectively. Occasional temperatures below freezing have been recorded during all seasons of the year. The growing season is about 190 days in length.

During this period, frosts occur 25 cm above the ground during an average of 35% of the nights. The annual average precipitation recorded is around 800 mm during the period 1988-1999. The recorded amounts compare well with those registered at a nearby meteorological observing station run by the Swedish Meteorological and Hydrological Institute. The annual average evapotranspiration at Asa is close to 500 mm and the calculated average runoff of water from the forest area is about 350 mm yr⁻¹. About 25% of the precipitation falls as snow. On the average, snow starts to fall on the 20th of November and the snow cover lasts until the beginning of April. Between-year variations are large and occasionally the snow cover can be almost absent. The Asa case-study area has a relative relief of up to 125 m, and the highest parts being found at about 285 m a.s.l. The Asa area contains a large number of lakes which, generally, are elongated in a north-south direction and are often separated by marked ridges.

The climate in Asa has changed during the last decade to being slightly milder and having less permanent snow. The present forest management is focused on optimizing production of Norway spruce, little attention being paid to deciduous trees. At present, the average production in Norway spruce stands is 9.1 m³ stemwood ha⁻¹yr⁻¹, and in Scots pine 5.5 m³ stemwood ha⁻¹yr⁻¹. The total area of productive forest in the case study area is 2,871 ha. The forest is relatively productive, the mean productivity being 8.1 m³ha⁻¹yr⁻¹. The highest productivity is found on the western and south-eastern parts of the area. The mean standing volume per ha is 128 m³ha⁻¹, Norway spruce being the most common tree species (89 m³ha⁻¹). The total volume in the case study area is ca 370,000 m³. Most of the forest is relatively young. In total, 70% of the forest is less than 40-years old and 30% is less than 20-years old. Old growth is rare, only 1.1% of the forest being above 120 years and 0.6% above 140 years of age.

The dominating silty-sandy till with low base saturation is reflected in the ion composition of the ground- and surface water. The mineral soil is often acidified to more than 1 m in depth. The typical chemistry of both soil and runoff water is low pH (4.5-5.5), low concentrations of base cations and inorganic nitrogen, and high concentrations of aluminium (0.01-0.03 mg l⁻¹). The small streams are more or less brown coloured by organic matter from the surrounding forest soils and peatlands. An intensive study of soil water in one spruce stand showed a large variation in element concentrations (Akselsson and Westling, 1999). The study indicated that the small-scale variation can be almost as large as the variation in the entire Asa area. The present concentrations in soil water and runoff are influenced by decades of high deposition of acidity. The response of water to the decreasing acid deposition is however delayed due to slow soil processes. The bedrock under the Asa area consists of old igneous granites (Smålandsgranit), dominated by quartz, K-feldspars and plagioclase, there being small amounts of biotite and hornblende in the granite. The area has small old intru-



Figure 13.1: *Asa, located in the middle of the province of Småland, is typical of a landscape once dominated by mixed forests and deciduous trees, but at present having primarily conifers, 80%. The contour is flat, the landscape being characterized by low rolling hills and thousands of lakes and small streams.*

| | | |
|------------------------|---------|---------------------------------|
| Land area | 3288 | ha |
| Productive forest area | 2871 | ha |
| Total volume | 368,200 | m ³ |
| Spruce | 255,500 | m ³ |
| Pine | 94,300 | m ³ |
| Birch | 13,800 | m ³ |
| Volume | 128 | m ³ ha ⁻¹ |

Table 13.1: *Data on the forests in Asa forest*

| Forest type | Area | Productivity, |
|-----------------------------|----------|--|
| Deciduous (>30 % broadleaf) | 14 ha | 3.3 m ³ ha ⁻¹ yr ⁻¹ |
| Deciduous | 108 ha | 4.4 m ³ ha ⁻¹ yr ⁻¹ |
| Mixed | 175 ha | 8.1 m ³ ha ⁻¹ yr ⁻¹ |
| Pine | 338 ha | 5.5 m ³ ha ⁻¹ yr ⁻¹ |
| Mixed conifer ha | 596 ha | 7.3 m ³ ha ⁻¹ yr ⁻¹ |
| Spruce | 1,562 ha | 9.1 m ³ ha ⁻¹ yr ⁻¹ |
| Clear-cut | 79 ha | 8.9 m ³ ha ⁻¹ yr ⁻¹ |
| Sum/Mean | 2,871 ha | 8.1 m ³ ha ⁻¹ yr ⁻¹ |

Table 13.2: *Area and productivity of the forest types in Asa forest*

sions of porphyrites and gabbroitic diabases (dominant minerals are epidote, pyroxene, garnet and hornblende). The presence of significant amounts of epidote and of almost no pyroxenes suggests the intrusions to have been subject to considerable hydrothermal alterations. The soil material at Asa was formed during the late ice-age and was laid bare 14,000 years ago. The soil mineralogy is estimated to have originated from a northwardly focused oval approximately 20 km in diameter. The stone and boulder content of the till is estimated to represent 30-60% of the volume.

13.2 Introduction

A landscape projection model is developed to illustrate the consequences of forest management for the local economic output, the base cation budget, the nitrogen leaching and the biodiversity. A number of management plans were made and certain aspects of the outcome were predicted and evaluated. The management plans are based on a sequence of steps according to Fig. 13.2. The properties of the landscape - physical, chemical, biological and socio-economic - are combined with a set of stand management programs and land management approaches to develop management programs. The management programs are input to the third component, the landscape projection model. Within the landscape projection model physical, chemical and biological processes and economic results are modelled. Values for indicators or for criteria are predicted for each management plan by use of the planning model and are evaluated with respect to profit and sustainability aspects. The planning procedure is initialised by applying a strategic set of landscape management programs and repeated until the outcome of management matched the objectives sufficiently well.

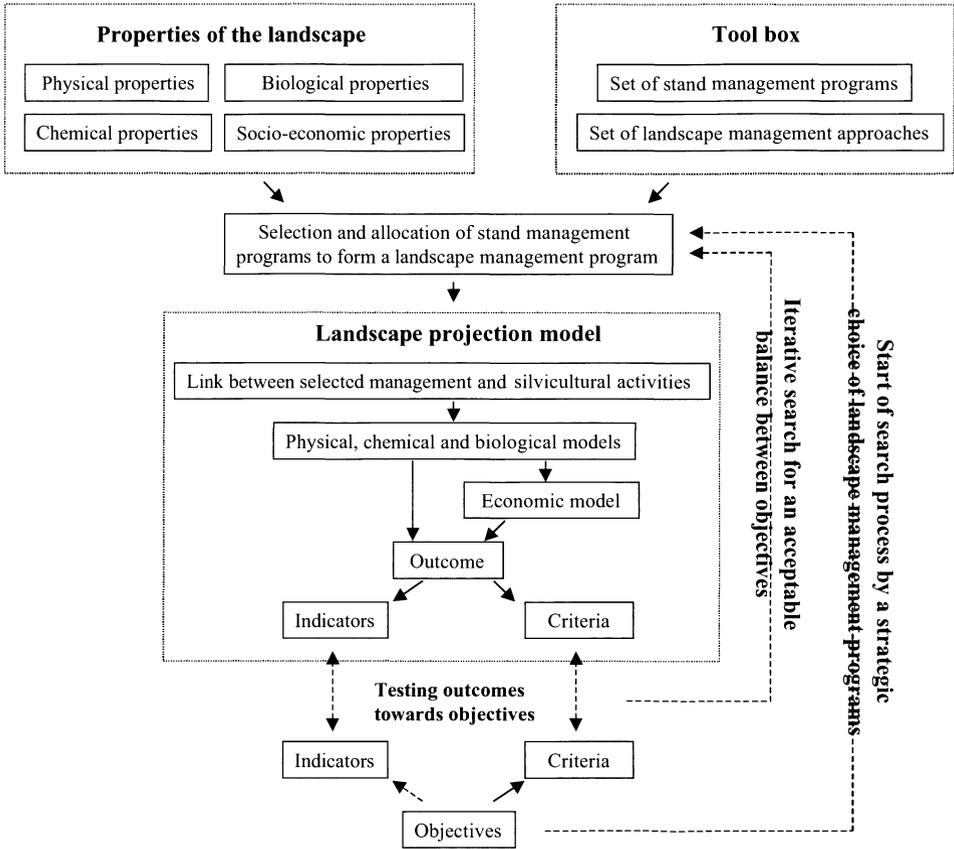


Figure 13.2: Flow of events in the planning model.

13.3 The landscape model

The landscape projection model used in the study is time-discrete and treats the stand as the smallest geographical unit. The model was developed for studying the long-term consequences of forest management programs (Fig. 13.2). Management decisions are treated as external variables. The frost hazard and windthrow models described earlier were used in the landscape model.

13.3.1 The tree cover model

The tree cover model uses the stand as the smallest geographical unit. Within a stand the trees are recorded in three layers: the production layer, the shelter-wood layer and the old-growth layer. In a stand, trees can belong to any of the three layers, or be found in all three layers simultaneously. Within each layer, the tree species pine, spruce, birch, beech, oak, and unspecified species as a group, are separated. The tree cover model has three development phases, where different biological processes are modelled in each phase:

1. The young forest phase, the period from regeneration to 8 m height

2. The production phase, the period between 8 m height and final felling;
3. The old-growth phase, defined as the period between the final felling and the final decay of the wood.

The model uses stand data that describes site conditions (site index, vegetation type, ground moisture, frost), stand characteristics (age, basal area, number of trees, diameter), history of silvicultural treatment (years since last thinning) and geographical location (links to neighbouring stands). During the young-forest phase regeneration and growth up to a tree height of 8 meters occur. The main processes/variables are the numbers of regenerated stems, the time from final felling until the trees enter the production phase, and the calculation of basal area when the trees enter the production phase. The total number of stems after regeneration is a function of the number of seedlings planted and the number of naturally regenerated seedlings. The number of plants is recorded according to tree species. The number of seedlings planted depends on the management program selected (0-3,500 years). The number of naturally regenerated seedlings of a given tree species is a function of its ability to disperse to the surrounding and the representation of the tree species:

1. within the stand,
2. in neighbouring stands,
3. in the landscape

The number of naturally regenerated seedlings is adjusted in relation to whether ground preparation has taken place, the amount of shading by shelter-wood, the site-index and the ground moisture (Hagner 1962, Karlsson 1994). The dependent variable is the number of seedlings obtained of a tree species at 1.3 meters. The function is estimated from several regeneration experiments, mainly in southern Sweden (Hagner 1962, Karlsson 1994). The number of seedlings is

$$n_{Regenerated} = n_{Planted} + n_{Natural} \quad (13.1)$$

The number of naturally regenerated seedlings is calculated as

$$n_{Natural} = (A_{stand} \cdot B_{stand} + A_{neighbor} \cdot B_{neighbor} + A_{landscape} \cdot B_{landscape}) \cdot C \cdot D \cdot E \cdot F \quad (13.2)$$

$$A_i = a \cdot (1 - b^{Basal\ area_i})^c \quad (13.3)$$

where A is the number of seedlings of a given tree species at 1.3 meters as a function of the basal area of that tree species within the stand (A_{stand}), within the neighbouring stands ($A_{neighbors}$), and the landscape ($A_{landscape}$). a, b, and c are fitted parameters, B is a dispersal coefficient (0-1), C is the correction if ground preparation is not carried out (0-1) (1 if ground preparation is carried out), D is the correction for the basal area in shelter-wood (0-1), E is the correction for site-index being low, medium or high (0-2) (1 for medium site-index), F is the correction for ground moisture being dry, mesic or wet (0-2) (1 for mesic). i is stand, neighbour or landscape. "Basal area stand" is the basal area in shelter-wood layer and old growth layer within the stand "Basal area neighbours" is the average basal area in production layer, shelter-wood layer and old growth layer in the neighbouring stands. "Basal area landscape" is the average basal area in production layer, shelter-wood layer and old growth layer in the landscape. The time it takes from the final felling until the trees enter the production phase is a function of the base time, defined as the time it takes for a 2-year-old plant, planted one year after a clear-cut, to become 8 meters high, this varies between 14 and 61 years, depending on the site-index and the tree species (Bergman 1987, Carbonnier 1971, Carbonnier 1975). A time delay is added to the base time in the event of planting on patches with increased relative risk of frost during the growing season. In the case of natural regeneration under shelter-wood, a time delay is added to allow for the waiting period for seed years to arrive, the time for germination and establishment, and

reduced growth height due to shading by shelter-wood. The time for trees to reach 8 meters in height is calculated as

$$t_{8\text{meters}} = t_0 + D_1 \cdot t_{\text{Frost}} + D_2 \cdot (t_{\text{Germination}} + t_{\text{Establishment}} + t_{\text{Shading}}) \quad (13.4)$$

where D_1 takes the value 1 if the stand is clear-cut otherwise 0, D_2 has the value 1 if natural regeneration is used otherwise 0, t_{Frost} is the time delay for frost injuries dependent of the relative frost risk during the growing season, $t_{\text{Germination}}$ is the the average time between seed years, Establishment is the time it takes for germinated seedlings to become the size of a planted seedling, t_{Shading} is the time delay due to shading from shelter-wood. The functions used to calculate the basal area during the transition from the young forest phase to the production phase are taken from Elving and Hägglund (1975) for pine, from Eriksson (1976) for spruce, from Johansson (1998) for birch, from Carbonnier (1971) for beech, and from Carbonnier (1975) for oak. The functions have different mathematical forms, but are all based on the independent variables of tree height and total stem number.

The growth simulator used for the production phase was developed within the HUGIN-project (Lundström and Söderberg 1996, Ekö 1985). This is an empirical system for the long-term forecasting of forest yield, based on a national growth database. HUGIN has been used on several occasions during the last two decades to simulate the effects of different forestry policies on future wood supply in Sweden (Svensson 1980). The main advantage of using this material instead of material from permanent research plots was that it is extensive and representative of Swedish forests. The main disadvantage was that at that



Figure 13.3: *A certain fraction of the area can be set aside for biodiversity conservation. Under sustainability management, such fractions can be located to areas of low potential for production or to areas with poor access, to minimize the impact on profitability. (Photo: Mats G. E. Svensson)*

time the NFI (National Forest Inventory, Riksskogstaxeringen) was only based on temporary plots. Therefore, only a brief description of the stand history was possible. Growth had to also be calculated from cores extracted from sample trees. The simulator consists of three sets of regression functions: basal area growth functions, volume functions, and functions to estimate mortality. The basal area growth functions are the most important functions for the simulator. The growth model was based on the assumption of Baule (1917), that the effects of different growth factors interact multiplicatively. On the basis of this assumption, a regression model was derived that could be parameterized using data from the NFI. The dependent variable was basal area growth during a 5-year period. Basal area growth was preferred to volume growth, since volume growth also includes a change in bole shape and is therefore assessed with less accuracy. Different functions were estimated for different species, different geographical regions, different site indices and different thinning histories (thinned, unthinned). The NFI contains data from both monocultural and mixed forests. The dependent variable therefore refers to the

basal area of a certain species within the plot and not to the basal area of the sample plot. In case of forecasting mixed forest a separate growth function has to be used for each of the existing species. The independent variables can be divided into those that describe site conditions (site index, vegetation type, latitude), stand characteristics (age, basal area, number of trees, basal area of competing species) and stand history (number of years since the last thinning). For longer forecasts, growth functions are applied recursively for one 5-year period at a time. Volume functions are used to estimate the stand volume at every five-year interval, based on the updated basal area and other stand and site characteristics. Dominant height is usually included in volume functions but could not be used, since height curves are only available for pure stands. The mortality functions refer to two types of mortality. The first type is caused by competition and the second by storm, snow-breakage, etc. The latter type does not include the effects of major incidents, such as extensive windthrows. The independent variable is the proportion of the basal area that dies during the 5-year period. A forecast will include the following steps:

1. Estimation of the five year basal area growth for each species, based on site conditions, stand characteristics and stand history at time t_0
2. Calculation of volume at $t_0 + 5$ years
3. Estimation of mortality during t_0 to $t_0 + 5$ years
4. Decision if thinning should be carried out
5. Updating of variables describing the stand at $t_0 + 5$ years
6. Estimation of five-year basal area growth for each species, based on site conditions, stand characteristics and stand history at time $t_0 + 5$

Validation of the growth simulator, by use of other simulators and of time series from permanent plots yielded satisfactory results. The accuracy of the forecast is complicated to investigate due to that both genuine and statistical errors will occur. The genuine errors are caused by factors in the growth simulator that are not under control, e.g. climatic change, nutrient deficiencies and environmental pollution. The statistical errors can be divided into sample errors, specification errors and errors of measurement. Specification errors can lead to large deviations especially when extrapolations occur. It is important, therefore, to interpret the results from simulations of uncommon forest types and from silviculture with care. A calculation was made of the relative standard error of the total growth in a stand as obtained from ten sample plots (Holm 1981). For the first five years the error was 17%. For a 50-year period it was reduced to 9%. However, for still longer periods there was only a minor further reduction in standard error. In situations where no extrapolation errors occur the most important source of deviation is the climate.

The old-growth layer reduces growth in the production layer and the shelter-wood layer. In order to reduce growth in the model the production layer and the shelter-wood layer were restricted to the production area. The production area is defined as the area that is not occupied by old-growth layer. The denser the old-growth layer is the smaller the production area becomes. The production area is calculated as a function of the basal area of the old-growth layer (Fries 1964, Carbonnier 1971, Carbonnier 1975, Eriksson 1976, Agestam 1985). The production area is recalculated each time a final felling occurs.

$$Production\ area = 1 - a \cdot Basal\ area \quad (13.5)$$

such that

$$Production\ area \geq 0 \quad (13.6)$$

where *Basal area* is the basal area of the old-growth layer and *a* is a parameter (0.029-0.05). When no old-growth layer is present, the production area is assigned the value of 1.0, implying

| DBH (cm) | Pine | Spruce | Birch | Beech | Oak | Other |
|----------|------|--------|-------|-------|-----|-------|
| 20-39 | 25 | 15 | 10 | 10 | 20 | 10 |
| 40-69 | 35 | 25 | 15 | 15 | 30 | 15 |
| 70+ | 55 | 35 | 20 | 20 | 45 | 20 |

Table 13.3: *Decomposition of dead trees. Average residence time (in years) in each decomposition class. For each diameter class, the three decomposition classes have the same residence times.*

that the entire stand area is available as the production layer and the shelter-wood layer. As soon as there are any old-growth trees, the production area is reduced according to Eq. 13.5. When the production area becomes 0, no area in the stand is available any longer for the production layer or the shelter-wood layer. In the forest, tree growth slows down considerably as trees become older and enter the "old-growth stage". At an age 1.2-2 times over the normal felling age (70-120 years), most tree species gradually enter a stage in which they begin to host a larger number of species, mainly epiphytic lichens and insects (Nilsson 1997, Thorén 1997, Fahlvik 1999). Many red-listed species are dependent on old trees, due to the formation of rough bark, large dead branches, hollows and other features not found in younger trees (Nilsson 1997, Nilsson et al. 2000). For biodiversity, central processes in the natural forests are also the formation of dead wood and of gaps that produce large microclimatic variations (Kaila et al. 1997, Nilsson 1997, Palm 1959, Arup et al. 1999, Berg et al. 1994). Diameter growth is species-specific, both with regards to growth level and to growth rhythm, although in practice no data on the growth dynamics of European tree species is available, so the diameter growth model is based on the "normal" maximum diameter and maximum ages recorded in ongoing studies and fieldwork in southern Sweden (Fahlvik 1999). Nesting trees is the collective name for trees that have attained the age at which they normally host a larger number of accompanying species, mainly epiphytic lichens, beetles and butterflies, due to bark characteristics, presence of dead wood, large branches, etc. (Berg et al. 1994, Jonsell et al. 1999, Jonsell et al. 1998). The term refers to the possible presence of holes and cavities for insects such as *Osmoderma eremita* (Ranius and Nilsson 1997) and for birds (Amcoff and Ericsson 1996). At higher tree ages a large number of rare fungi species may also be found (Bader et al. 1995, Jonsell 1999). Although the time when "nesting trees" start to develop is species-specific and is also dependent on the stand structure and stand history (Nilsson 1997, personal communication, Niklasson), a higher age generally means a greater probability that a single tree will become a "nesting tree". The life span is species-specific (Schweingruber 1992) and resembles very much the principle of nesting-tree formation, but mortality usually takes place after the tree has entered the "nesting tree" stage. However, a tree can also die before reaching this stage, typically as a result of strong winds or severe insect attacks. Dying is modelled by the probability of a single tree dying. The quality of dead wood in terms of promoting biodiversity (mainly wood-living beetles and fungi) depends upon its original dimension, species, sun exposure and degree of decomposition, i.e. how long it has been dead (Falinski 1986, Harmon et al. 1986, Krankina 1999). Because of lack of available data on the decomposition of dead wood for the deciduous species in Europe (Kirby et al. 1998), mainly unpublished data and field observations was used as a basis for assigning a rate of decomposition, supporting this with dendrochronological cross-dating of tree-remains. The decomposition model for dead wood includes three diameter classes and three decomposition classes (Tab. 13.3).

13.3.2 Nutrient budgets and nitrogen leaching

The method employed to model nutrient budget and nitrogen leaching involves making a budget calculation for each cation separately. Taking deposition and weathering as the sources of base

| Deposition kEq/ha | Water | H+ | SO4 | Cl | NO3 | NH4 | Ntot | Ca | Mg | Na | K | Ali |
|-------------------------------------|-------|-------|-------|-------|-------|-------|-------|-------|-------|-------|-------|-------|
| Spruce>60 yr | 444 | 0.330 | 0.424 | 0.437 | 0.353 | 0.351 | 0.811 | 0.126 | 0.109 | 0.372 | 0.052 | <0.01 |
| Spruce 50-60 yr. pine>60 yr | 522 | 0.308 | 0.391 | 0.423 | 0.336 | 0.336 | 0.779 | 0.119 | 0.108 | 0.359 | 0.050 | <0.01 |
| Spruce 30-50 yr. pine 30-50 yr | 579 | 0.289 | 0.361 | 0.423 | 0.321 | 0.321 | 0.750 | 0.114 | 0.105 | 0.361 | 0.048 | <0.01 |
| Shelter-wood | 743 | 0.259 | 0.325 | 0.423 | 0.307 | 0.307 | 0.721 | 0.114 | 0.105 | 0.361 | 0.050 | <0.01 |
| Deciduous. spruce and pine 10-30 yr | 619 | 0.250 | 0.325 | 0.366 | 0.293 | 0.293 | 0.693 | 0.113 | 0.099 | 0.313 | 0.047 | <0.01 |
| Forest<10 yr | 743 | 0.240 | 0.313 | 0.338 | 0.286 | 0.286 | 0.679 | 0.111 | 0.095 | 0.291 | 0.046 | <0.01 |
| Clear-cuts | 825 | 0.216 | 0.286 | 0.288 | 0.272 | 0.270 | 0.649 | 0.109 | 0.081 | 0.256 | 0.045 | <0.01 |
| Rooting zone kEq/ha | Water | H+ | SO4 | Cl | NO3 | NH4 | Ntot | Ca | Mg | Na | K | Ali |
| Spruce>60 yr | 200 | 0.033 | 0.424 | 0.437 | 0.005 | 0.003 | 0.115 | 0.104 | 0.101 | 0.404 | 0.006 | 0.257 |
| Spruce 50-60 yr. pine>60 yr | 250 | 0.031 | 0.391 | 0.423 | 0.005 | 0.003 | 0.115 | 0.099 | 0.096 | 0.384 | 0.006 | 0.244 |
| Spruce 30-50 yr. pine 30-50 yr | 300 | 0.030 | 0.361 | 0.423 | 0.005 | 0.003 | 0.115 | 0.096 | 0.093 | 0.371 | 0.006 | 0.236 |
| Shelter-wood | 450 | 0.029 | 0.325 | 0.423 | 0.005 | 0.003 | 0.115 | 0.091 | 0.089 | 0.355 | 0.006 | 0.226 |
| Deciduous. spruce and pine 10-30 yr | 350 | 0.027 | 0.325 | 0.366 | 0.005 | 0.003 | 0.115 | 0.085 | 0.082 | 0.330 | 0.005 | 0.210 |
| Forest<10 yr | 450 | 0.025 | 0.313 | 0.338 | 0.005 | 0.003 | 0.115 | 0.080 | 0.078 | 0.312 | 0.005 | 0.198 |
| Clear-cuts | 500 | 0.060 | 0.286 | 0.288 | 0.500 | 0.009 | 0.652 | 0.148 | 0.113 | 0.228 | 0.095 | 0.416 |
| Catchment kEq/ha | Water | H+ | SO4 | Cl | NO3 | NH4 | Ntot | Ca | Mg | Na | K | Ali |
| Mixed stands | 350 | 0.059 | 0.346 | 0.385 | 0.002 | 0.001 | 0.110 | 0.156 | 0.136 | 0.337 | 0.014 | 0.056 |

Table 13.4: Calculated annual water flux, present deposition and future areal loss of different elements in steady state at the current deposition (mean of 1996-1999) of sulphur and chloride, $kEq ha^{-1} yr^{-1}$.

cations, and uptake and leaching as the sinks, the mass balance for each base cation becomes (Sverdrup and Rosen 1998):

$$DELTA = U - D - W + L \quad (13.7)$$

where U is uptake, which depends on the amounts removed from the system by harvesting, D is deposition, which depends partly on human activities, W is weathering and L is leaching, which depend on water flux and acid deposition. In principle, this limitation applies to any nutrient, as well as to water. Such a mass balance can be made for any bulk or trace nutrient. Sustainability is maintained when the long-term average value of DELTA is zero. Budget calculations were made for Ca, Mg and K. The amount of biomass harvested is calculated using the tree cover model. The biomass extracted is quantified in terms of dry weight by use of regression models estimated from data presented in Marklund (1987) (Pettersson 1999). The biomass is separated into three fractions: stem wood and bark, branches and needles. The amount of nutrients removed was calculated by multiplying the micronutrient per unit content by the amount of biomass extracted. The micronutrient content according to Egnell et al. (1998) was used. For other tree species than pine, spruce and birch, the micronutrient content for birch was used. Calculations of water fluxes, current deposition and future areal loss of different elements were made for various forest types. Fluxes for runoff are based on data from acidified streams. The calculations of element fluxes assume a long-term balance between the input and output of both SO_4 and Cl. It is also assumed that the current base saturation in the mineral soil will remain relatively unchanged during the coming decades. This implies that the soil- and surface water will also be acidic in the future and show a distribution of elements like the current water chemistry. The deposition and the areal loss are dependent on the characteristics of the forest stand. Dry deposition increases with an increase in leaf area index, the height of the trees and number of stems in the stand. Wet deposition is more evenly distributed over forest and open areas. The calculated mean N deposition in the area varied from $9 kg ha^{-1} yr^{-1}$ in open field to $11 kg ha^{-1} yr^{-1}$ in old spruce forests during 1996-1999. A large part of the Mg deposition, and to some extent also of Ca and K, originates from sea salt.

The following empirical regression model was estimated from Table 13.3.2 and used in the projection model to calculate deposition:

$$D = a_1 \cdot B + b_1 * H + k_1 \quad (13.8)$$

where D is deposition of each cation in $\text{kEq ha}^{-1}\text{yr}^{-1}$, B is basal area (m^2), H is average tree height (m), a and b are parameters and k is a constant. The following models, estimated from Table 3.3 were used in the projection model to calculate areal loss:

$$L_1 = a_2 \cdot B + b_2 \cdot H + k_2 \quad (13.9)$$

where L_1 is areal loss in $\text{kEq ha}^{-1}\text{yr}^{-1}$, a_1 and b_1 are parameters, and k_1 is a constant.

$$L_2 = (a_3 - b_3 \cdot B) \cdot D_3 \cdot (1 + (D_4 \cdot 0.2 - D_5 \cdot 0.2)) \quad (13.10)$$

so that

$$(a_3 - b_3 \cdot B) \geq 0 \quad (13.11)$$

where L_2 is areal loss in $\text{kEq ha}^{-1}\text{yr}^{-1}$ during the regeneration phase, a_3 and b_3 are parameters, D_3 is a variable that takes on the value of 1 if final felling has been carried out during the last five years, and otherwise 0, D_4 is a variable that takes on the value of 1 if final felling and scarification is carried out, and otherwise 0, and D_5 is a variable that takes on the value of 1 if final felling and slash removal is carried out, and otherwise 0.

$$L = ABR_1 \cdot L_1 + ABR_2 \cdot L_2 \quad (13.12)$$

where L is the total areal loss in $\text{kEq ha}^{-1}\text{yr}^{-1}$, and ABR_1 and ABR_2 are relations between total anion leaching and specific base anion leaching for K, Ca and Mg. The concentration of inorganic nitrogen in the soil water at 0.5 m depth follows a pattern typical after clear-cutting. Usually a peak in leaching occurs ca 3 years after cutting. Areal loss has been calculated from clear-cuttings of mixed conifer forests on typical till soils at the Asa Experimental Forest. The concentrations usually peak at about $1.5\text{-}2 \text{ mg l}^{-1}$. The total areal loss during the five-year period after clear-cutting is $30\text{-}40 \text{ kg ha}^{-1}$. Treatments that have an influence on inorganic nitrogen leaching are shelter-wood, slash removal and scarification. In some experiments conducted on mineral soils in conifer forests, shelter-wood have been found to be effective in preventing nitrogen leaching. By using ca $150 \text{ stems ha}^{-1}$ it was possible to decrease leaching to the conditions in the surrounding forest. The stem density is crucial for the effect of shelter-wood. In a study performed in a number of stands located in a gradient from Asa to the west coast of Sweden, virtually no inorganic N leaching was found at shelter-wood densities above ca $12 \text{ m}^2 \text{ ha}^{-1}$. Lower stem densities reduced the leaching of N as compared with clear-cuts. Based on this observation a model in which leaching was linearly related to stem density up to $12 \text{ m}^2 \text{ ha}^{-1}$ was estimated. Slash removal tends to decrease the leaching of nitrogen, although the effect is not very strong. In the model, the effect of slash removal was estimated to be a 20% reduction in the leaching of nitrogen. Scarification was also found to have some effect on the leaching of nitrogen, although the effect was smaller than what had previously been assumed for Sweden. In some experiments scarification has not at all affected nitrogen leaching, whereas other experiments show some increase. In the model, the effect of scarification was estimated to be a 20% increase in the leaching of nitrogen. The following equation was used:

$$N_{leach} = (a - b \cdot B) \cdot D_3 \cdot (1 + (D_4 \cdot 0.2 - D_5 \cdot 0.2)) \cdot D_6 + c \quad (13.13)$$

so that

$$(a - b \cdot B) \geq 0 \quad (13.14)$$

where B is basal area of the shelter-wood, a , b and c are parameters, D_3 is assigned the value of 1 if final felling has been carried out during the last five years, and otherwise 0, D_4 is assigned the value of 1 if final felling and scarification have been carried out, and otherwise 0, D_5 is assigned the value of 1 if final felling and slash removal have been carried out, and otherwise 0, and D_6 is assigned the value of 3 if prescribed burning has been carried out, and otherwise 0.

| Item | Scarified | Not scarified |
|--------------|-----------|---------------|
| Shelter-wood | 0.035 | 0.015 |
| Clear-cut | 1.037 | 0.859 |

Table 13.5: Mean concentration of inorganic nitrogen ($[NO_3^-] + [NH_4^+]$ mg N t^{-1}) 1-4 years after cutting. Plots were scarified the first autumn after cutting. The shelter-wood density was about 150 stems ha^{-1} , dominated by Scots pine. The mean for three sites, Lönsboda, Asa and Siljansfors, all with till soils and mesic soil moisture, are given

13.4 Indicators

It should be pointed out that biodiversity assessment using only structures, employed here, is still of unknown reliability (Nilsson et al. 2000). In practical tests, monitoring of indicator species in addition to structures is mandatory (Nilsson et al. 2000). Based on the strategy presented above, the following indicators are used for assessing biodiversity:

1. Densities of living trees of different dimensions, deciduous trees with a DBH above 70 cm.
2. Densities of dead trees of different dimensions, dead deciduous trees with a DBH above 40 cm.
3. Amount and supply of recently burnt trees.
4. Spatial distribution of living and dead trees

The mass balance for base cations consist of atmospheric deposition and weathering (three rates) minus biomass removal by harvest (calculated from the management programs) and leaching. The criteria used for assessing the mass budget of base cations were

1. Base cation deficit
2. Leaching of nitrogen from the root zone.

The criteria used for assessing wood production and economic return is focused on the growing stock:

1. Changes in mean volume of the growing stock on forest land (non-declining).
2. Changes in diameter distribution of the growing stock (non-declining).

Wood production

1. Growth (non-declining).
2. Difference between growth and removals of wood (>0).

Wood extraction

1. Removal of wood (non-declining).
2. Percentage of timber of removals (non-declining).

Resulting micro-economy of wood extraction

1. Changes in net return (non-declining).
2. Average net return over the simulation period (SEK/ha).

13.5 Forest management

Selection of the set of stand management programs to be used in designing a landscape management program is based on three considerations:

1. The properties of the landscape in term of its physical, chemical, biological and socio-economic limits.
2. The objectives defining the direction in which the stand management programs should aim.
3. The silvicultural principles and experience available, in terms of regeneration strategies, thinning regimes and rotation length

Formulating a set of stand management programs is a mixture of the three components where (1) and (2) define the area within which feasible stand management programs can be found and precisely defined according to (3). The set of stand management programs was decided upon and that was utilised then in designing the landscape management programs is described in detail below. The stand management programs can be divided roughly into three categories: Wood production, Water quality and Biodiversity. For each of these categories a number of specific programs are available. It is possible to combine the different management programs such as Wood production and Biodiversity to produce both timber and biodiversity within a given stand.

13.5.1 Stand management programs

The stand management program *Wood production*

The stand management programs that aim to wood production of Scots pine, Norway spruce and mixed conifer stands will use natural regeneration if the site index is less than T24 or if the soil moisture class is wet. In all other cases, regeneration is accomplished by planting. Planted seedlings are treated against browsing by roedeer and moose and against damage by pine weevils. If the site index is below 26 m for pine, and below 28 m for spruce, and if the basal area of Scots pine in the previous stand was larger than $6 \text{ m}^2 \text{ ha}^{-1}$, a shelter-wood is left after final cutting in order to protect the planted seedlings and provide seeds for complimentary natural regeneration. The basal area of the shelter-wood is $12 \text{ m}^2 \text{ ha}^{-1}$ and Scots pine will have priority as shelter-wood-trees. A shelter-wood is always e left on regeneration areas on which the risk of frost damage is high. The shelter-wood is cut 5-9 years after the start of regeneration. Scarification is done in all cases except on wet sites. Wet sites are drained and very wet sites regenerate natural (shelter-wood of $15 \text{ m}^2 \text{ ha}^{-1}$) Scots pine, Norway spruce and mixed conifer stands are thinned pre-commercially 5-9 years after being established. In both Scots pine and Norway spruce, 3-4 thinnings are done (from below), 25-30% of the basal area being removed in each thinning. The clear-cut age is dependent on site index, and varies between 65-124 years for Norway spruce and 80-129 years for Scots pine.

The stand management program that aims to produce birch makes use of natural regeneration after scarification if birch was present in the previous stand. If possible, the seed-trees of birch should have a basal area of $3 \text{ m}^2 \text{ ha}^{-1}$, and all birch trees are left as seed-trees if the basal area of birch before final felling was less than $3 \text{ m}^2 \text{ ha}^{-1}$. The seed-trees are cut 5-9 years after the start of regeneration. If no birch were present before the final felling, seedlings are planted. Before planting, the regeneration area is scarified and if the area is wet, drainage is carried out. If the percentage of coniferous trees in the stand before the final felling is above 70%, the birch seedlings are protected against pine weevil damage. If the planting of broadleaves is not permitted, the regeneration area is clearcut and scarified, the birch seeds transported by wind from adjacent stands being used for natural regeneration. The birch stands are thinned pre-commercially twice, the first after 5-9 years and the second after 10-29 years, dependent on

| Site index | Pre-thinnings | | | | | Thinnings | | | | | Final felling age | | | | |
|------------|---------------|----|----|-------|-------|-----------|-------|--------|--------|--------|-------------------|-------|--------|--------|-----|
| | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | | 11 | | | |
| Spruce | | | | | | | | | | | | | | | |
| G16 | 9 | | | 54:25 | 74:25 | 104:20 | | | | | | | 124 | | |
| G24 | 9 | | | 44:30 | 64:25 | 84:20 | | | | | | | 109 | | |
| G36 | 9 | | | 24:30 | 34:30 | 44:25 | 54:25 | | | | | | 69 | | |
| Pine | | | | | | | | | | | | | | | |
| T16 | 9 | | | 54:30 | 79:30 | 104:30 | | | | | | | 129 | | |
| T22 | 9 | | | 39:30 | 54:30 | 69:25 | 84:20 | | | | | | 104 | | |
| T30 | 9 | | | 29:30 | 39:30 | 49:30 | 64:30 | | | | | | 84 | | |
| Birch | | | | | | | | | | | | | | | |
| B16 | 9 | 29 | | 44:30 | 64:25 | | | | | | | | 74 | | |
| B22 | 9 | 19 | | 29:28 | 34:25 | 44:24 | 49:21 | | | | | | 64 | | |
| B28 | 9 | 14 | | 19:28 | 24:25 | 29:24 | 34:22 | 39:21 | | | | | 49 | | |
| Beech | | | | | | | | | | | | | | | |
| F16 | 9 | 24 | 34 | 64:20 | 74:20 | 84:20 | 94:20 | 104:30 | 119:30 | | | | 134 | | |
| F24 | 9 | 19 | 29 | 54:15 | 59:15 | 64:15 | 74:15 | 84:20 | 94:20 | 104:30 | 114:30 | | 129 | | |
| F32 | 9 | 14 | 24 | 44:15 | 49:15 | 54:15 | 59:15 | 64:15 | 69:15 | 74:20 | 79:20 | 89:20 | 99:30 | 109:30 | 119 |
| Oak | | | | | | | | | | | | | | | |
| E16 | 9 | 24 | | 49:21 | 59:33 | 69:30 | 79:28 | 89:26 | 99:33 | 114:23 | 129:21 | | 144 | | |
| E24 | 9 | 19 | | 39:15 | 44:23 | 49:20 | 54:20 | 59:18 | 64:17 | 74:27 | 84:22 | 94:19 | 109:22 | 124:20 | 139 |
| E28 | 9 | 19 | | 34:30 | 39:25 | 44:22 | 49:20 | 54:17 | 64:26 | 74:21 | 84:18 | 94:16 | 104:16 | 114:16 | 129 |

Table 13.6: *Stand age at the time of pre-commercial thinnings, thinnings and final felling for stand management program Wood Production. For thinnings the percentage thinned is also given. Stand age (years): Proportion of basal area (%) are given*

site index. The birch stands are thinned 3-5 times, 21-30% of the basal area being removed at each thinning. The rotation length depends on soil fertility, 45-49 years being used at the most fertile sites and 70-74 years at the least fertile ones. The stand management program that aims to produce beech uses natural regeneration if the basal area of beech in the stand prior to the final felling is more than $6 \text{ m}^2 \text{ ha}^{-1}$. A shelter-wood of $12 \text{ m}^2 \text{ ha}^{-1}$ is left if the site index for beech is less than 27 m. For higher site indices, $15 \text{ m}^2 \text{ ha}^{-1}$ of shelter-wood is left. The shelter-wood densities are reduced in three steps. If the site index for beech is below 22 m, the shelter-wood is finally removed after 25-29 years. For site indices of 23-27 m, the last shelter-wood is removed after 20-24 years, and for site indices of 28-36 m, the last shelter-wood is removed after 15-19 years. On dry and fresh sites, scarification is carried out and the beech-nuts are covered by mineral soil after seed-fall. If the basal area of beech in the previous stand was too small for natural regeneration, and if the user has specified planting of broadleaves as an option, planting is done after scarification. On sites with a high risk of frost damage, shelter-wood (basal area $12 \text{ m}^2 \text{ ha}^{-1}$) is left after cutting and wet sites are drained. If the site index for beech is above 23 m, the regeneration area is fenced in before planting, and if the previous stand contained more than 70% conifer species the seedlings are protected against pine-weevil damage. If a beech shelter-wood is impossible to establish and if the user has specified that planting of broadleaves is not allowed, the regeneration area is regenerated naturally by beech-nuts from adjacent stands. A shelter-wood ($12 \text{ m}^2 \text{ ha}^{-1}$) is left after cutting and dry-fresh sites are scarified. The beech stands are thinned pre-commercially three times. The first pre-commercial thinning is done 5-9 years after establishment of the stand the timing of the last two pre-commercial thinnings being dependent on site fertility. Six-eleven thinnings are done, dependent on site fertility, 15-30% of the basal area being removed in each thinning. The clear-cut ages vary from 115-119 years at the most fertile sites to 130-134 years at the least fertile ones. The stand management program that aims to produce oak uses plantation after scarification and field-vegetation control if the planting of broad-leaved species is specified as an option. At sites with a high risk of frost damage, shelter-wood is left to protect the planted seedlings. If the site index for oak is higher than 21 m, the regeneration area is fenced in before planting, on wet sites drainage will be done, the seedlings being protected against pine weevil

damage if the previous stand contained more than 70% coniferous species. If the planting of broad-leaved species is not allowed, oak is regenerated naturally. A shelter-wood ($8 \text{ m}^2 \text{ ha}^{-1}$) is left after the final felling, dry-fresh sites being scarified. The shelter-wood is cut in one step after 5-9 years. The oak is thinned pre-commercially twice, the first time after 5-9 years and the second time after 15-19 years at fertile sites and after 20-24 years at less fertile sites. Thinning is done 8-11 times, 17-33% of the basal area being removed at each thinning. The age of clear-cutting varies from 125-129 years at fertile sites to 140-144 years at less fertile ones. The stand management program that aims to produce other broad-leaved species (aspen, alder, etc.) regenerates in the following way: If the basal area of other broad-leaved species in the stand prior to the final felling is greater than zero, a shelter-wood ($5 \text{ m}^2 \text{ ha}^{-1}$) is left. Dry-fresh sites are scarified. If the previous stand did not contain any broad-leaved species except for birch, beech and oak, and if planting of broad-leaves is permitted, the tree-species desired is planted. Drainage is carried out on wet sites, the planted seedlings being protected against damage by pine weevil if the previous stand contained more than 70% coniferous species. The pre-commercial thinning, thinning and final felling in stands of other broad-leaved species follows the same program as for birch. The stand management program that aims to produce mixed broadleaf stands normally uses natural regeneration. If the combined basal area of beech and oak in the stand prior to the final felling is greater than $6 \text{ m}^2 \text{ ha}^{-1}$, shelter-wood ($12 \text{ m}^2 \text{ ha}^{-1}$) is left. Dry-fresh sites are scarified the nuts or acorns being covered by mineral soil. Wet sites are drained. If the basal area of oak and beech is too low for natural regeneration and if planting of broad-leaves is permitted, the stand is planted by the tree-species desired. If the risk of frost damage is high, 12 m^2 of shelter-wood ha^{-1} is left after cutting. The regeneration area is scarified prior to planting, wet sites are drained and, if the site index for beech is more than 23 m, the regeneration area is fenced. The planted seedlings are protected against damage by pine weevil if the previous stand contained more than 70% coniferous species. If planting of broad-leaves is not permitted, a shelter-wood ($12 \text{ m}^2 \text{ ha}^{-1}$) is left after cutting, dry-fresh sites being scarified and acorn and nuts transported from neighbouring stands being used for natural regeneration. The pre-commercial thinning, thinning and final felling of stands of noble broad-leaves follow the same program as for beech. The stand management program that aims to produce mixed broad-leaved stands uses natural regeneration if the basal area of broad-leaved species in the stand prior to the final felling is greater than zero. A shelter-wood is left after cutting ($8 \text{ m}^2 \text{ ha}^{-1}$). Dry-fresh sites are scarified. If the previous stand did not contain any broad-leaved species and if the planting of broad-leaves is permitted, the stand is planted by the desired tree species. Prior to planting, scarification is carried out and wet sites are drained. If the previous stand contained more than 70% coniferous species, the seedlings planted are protected against damage by pine weevils. If planting of broad-leaves is not permitted and if the previous stand did not contain any broad-leaved species, the stand is clear-cut and seeds transported from neighbouring stands to be used for natural regeneration. The pre-commercial thinning, thinning and final felling in stands of mixed broad-leaves follows the same program as for birch.

Forest Management Program *Water quality*

The main difference between the stand management programs and the minimised nitrogen leaching program is that in the latter a shelter-wood with a basal area of $12 \text{ m}^2 \text{ ha}^{-1}$ is left after final cutting, except for beech shelter-woods at fertile sites and coniferous stands at wet sites where $15 \text{ m}^2 \text{ ha}^{-1}$ shelter-wood are left. Pre-commercial thinnings, thinnings and final felling follow the same schedule as for wood-production.

Forest Management Program *Biodiversity*

For stands assigned to biodiversity, no regeneration measures are made except for areas assigned to "BIO-fire" which are treated with prescribed burning after clear-cutting. After clear-cutting,

| Site index | Pre-thinnings | | | Thinnings | | | | | | | Final felling age | | | | |
|------------|---------------|----|---|-----------|-------|-------|-------|--------|-------|---|-------------------|---|---|----|-----|
| | 1 | 2 | 3 | 1 | 2 | 3 | 4 | 5 | 6 | 7 | | 8 | 9 | 10 | 11 |
| Spruce | | | | | | | | | | | | | | | |
| G16 | 9 | | | 54:35 | 74:35 | 94:35 | | | | | | | | | 109 |
| G24 | 9 | | | 39:40 | 59:35 | 74:30 | 84:30 | | | | | | | | 94 |
| G36 | 9 | | | 24:40 | 34:40 | 44:35 | 54:35 | | | | | | | | 69 |
| Pine | | | | | | | | | | | | | | | |
| T16 | 9 | | | 54:30 | 69:20 | 84:20 | 99:15 | 114:15 | | | | | | | 149 |
| T22 | 9 | | | 39:30 | 54:25 | 69:25 | 84:15 | 99:15 | | | | | | | 134 |
| T30 | 9 | | | 29:25 | 39:25 | 49:25 | 64:20 | 79:20 | 94:15 | | | | | | 114 |
| Birch | | | | | | | | | | | | | | | |
| B16 | 9 | 29 | | 44:20 | 54:20 | 64:20 | 74:20 | 84:15 | | | | | | | 104 |
| B22 | 9 | 19 | | 29:20 | 34:20 | 44:20 | 59:15 | 69:15 | | | | | | | 89 |
| B28 | 9 | 14 | | 19:25 | 24:20 | 29:20 | 39:20 | 49:20 | 59:20 | | | | | | 74 |
| Beech | | | | | | | | | | | | | | | |
| F16 | 9 | 24 | | | | | | | | | | | | | 54 |
| F24 | 9 | 24 | | | | | | | | | | | | | 54 |
| F32 | 9 | 24 | | | | | | | | | | | | | 54 |
| Oak | | | | | | | | | | | | | | | |
| E16 | 9 | 24 | | | | | | | | | | | | | 54 |
| E24 | 9 | 24 | | | | | | | | | | | | | 54 |
| E28 | 9 | 24 | | | | | | | | | | | | | 54 |

Table 13.7: Stand age at the time of pre-commercial thinnings, thinnings, and final felling for the forest management programs "Biodiversity". The percentage thinned is also given for the thinnings. (Stand age (years): Proportion of basal area (%)).

the area is regenerated by seeds from trees retained from the previous stand or by seeds transported in from neighbouring stands, the desired tree-species being selected in pre-commercial thinnings. Stand management programs for biodiversity in mixed conifers, Scots pine and broad-leaves, differ from programs for the management of wood production by longer rotation periods and more frequent thinnings. Furthermore, the basal area removed in each thinning is less than for Wood Production. In Norway spruce, a larger proportion of the basal area is removed in each thinning, the rotation length being shorter. The stand management program that aims to produce noble broad-leaved stands only uses pre-commercial thinning. At the stand-age of 54-years, the final felling is done but all noble broad-leaved trees are retained in the final felling.

In the biodiversity programs, birch is handled in two different ways. BIO-Birch is handled as described above as regards to regeneration, thinnings, green tree retention, etc. In "WP-Birch long rotation" no trees are retained after the final cutting, regeneration treatments being as described for "WP-Birch" and pre-commercial thinning and thinnings are as described for "BIO-Birch". When the stands are clear-cut, 200 trees of the desired tree species are retained if possible. If the number of trees in the stand is low, all desired trees are retained. If the desired tree species are noble broad-leaves, all of the desired tree species are retained after clear-cutting.

13.5.2 Landscape management approaches

In this section we present briefly different approaches that can be used to form the landscape management programs needed to carry out model experiments with the landscape projection model (Fries et al. 1998). Landscape management has developed during the last decade, mainly as a result of new objectives in forestry (e.g. preserved biodiversity, aesthetics, recreation, etc.). A landscape perspective gives an opportunity to balance objectives in multiple-use forestry through allocating management activities within the landscape (e.g. management for nature conservation can be allocated to certain areas and wood production to others). The variety of approaches and models that have emerged for landscape management is a function of regional differences in land use history, forest conditions, recreation pressures, and ownership. Three approaches can be distinguished (Fries et al. 1998): the "species approach" and the "naturalness

| | MaxWP | MinN | BioCon | BioDisp |
|------------------------|-------|------|--------|---------|
| WP-Spruce | 1278 | | 998 | 1119 |
| WP-Pine | 301 | | 153 | 236 |
| WP-Mixedconifer | 1292 | | 985 | 982 |
| WP-Birch long rotation | | | 149 | 150 |
| H2O-Pine | | 301 | | |
| H2O-Mixedconifer | | 2570 | | |
| BIO-Pine | | | 2 | 7 |
| BIO-Birch | | | 130 | 63 |
| BIO-Beech | | | 5 | 5 |
| BIO-Oak | | | 173 | 31 |
| BIO-Fire | | | 278 | 277 |

Table 13.8: *The extent of stand management programs, expressed as stems ha⁻¹. WP = wood production, H2O = water quality, Bio = Biodiversity, Mixedconifer = mixture of spruce and pine, Birch long rotation = Birch with extended rotation compared with what is optimal in terms of productivity.*

approach” both of those integrate conservation aspects, and the “multiple aspects approach” which integrates several aspects (biological, social, economic, spiritual, etc.) in commercial timber-producing forestry. The “species approach” has two theoretical models that could be used to develop applicable models in practical forestry planning. These are island biogeography (MacArthur and Wilson 1967) and landscape ecology (Forman and Godron 1986). The “species approach” forms the basis for the landscape planning model termed “Key Habitat-Corridor Model”. This model is partly founded on island biogeography (MacArthur and Wilson 1967), but primarily on landscape ecology (Forman and Godron 1986). Island biogeography suggests species and gene dispersal between key habitats to be promoted if the key habitats are not too distant. It also suggests the probability that species survival in key habitats will increase with increasing key habitat size. Landscape ecology expands the theoretical basis for the model. This discipline emphasizes the spatial structure of entire landscape mosaics, the interactions between spatial elements, and changes in the spatial pattern over time. It deals with many of the shortcomings when island biogeographic theory is applied to terrestrial ecosystems, such as the fact that the matrix surrounding a key habitat in a forest landscape has a relatively complex spatial structure. A forest landscape matrix is also dynamic and ecologically important in itself (Franklin 1992, Liljelund et al. 1992, Oliver 1992, Wiens 1995). Combining different stand management programs within a given landscape forms a landscape management program. The number of possible landscape management programs is equal to the number of stands raised to the number of stand management programs. The number of possible landscape management programs becomes very large as soon as the number of stands increase (in our case approximately 2,500) or the number of stand management programs increases (in our case approximately 12). In order to limit the large number of possible landscape management programs, we formulated four that roughly pinpoint the boundaries of the others. These four are strategically chosen are “Maximise wood production” (MaxWP), “Minimise nitrogen leaching” (MinN), “Preserved biodiversity by concentrating nature conservation” (BioCon), and “Preserved biodiversity by dispersing nature conservation” (BioDisp). The concept of concentrated nature conservation is used to describe two things, first that nature conservation is the dominant objective in certain stands, and secondly that these programs are concentrated in the landscape, i.e. allocated close to each other. Dispersing nature conservation refers to the opposite, that nature conservation is integrated with wood production, and that these programs are applied

| | MaxWP | MinN | BioCon | BioDisp |
|--|--|-------------------------------|---|--|
| WP-Spruce | SI >T26/G27 | | SI >T26/G27 | SI >T26/G27 |
| WP-Pine | SI <T24/<G24 | | SI <T24/<G24 | SI <T24/<G24 |
| WP-Mixedconifer | SI >=T24- <=T26 and SI >=G24-<=G27 | | SI >=T24-<=T26 or SI >=G24-<=G27 | SI >=T24-<=T26 or SI >=G24-<=G27 |
| WP-Birch long rotation | | | Located close to Bio- programs and lakes | None of the other, BIOprograms and birch > 850 stems ha ⁻¹ |
| H20-Pine H20- Mixedconifer BIO-Pine | | SI <T24/<G24 SI>=T24/>=G24 | | |
| BIO-Birch | | | Reserve with Pine forest | Reserve with Pine forest |
| BIO-Beech | | | Located close to present Oak/Birch patches | Age > 50 years and birch > 50 stems ha ⁻¹ |
| BIO-Oak | | | All stands with Beech | All stands with Beech |
| BIO-Fire | | | Located close to present Oak patches | All stands with Oak and no Beech stems |
| | | | Located to an area with present Pine forest in the north east part | Pine>400 stems ha ⁻¹ , distance to road<100 m and SI<29 |

Table 13.9: *Criteria for allocating stand management programs in the landscape.* * T=pine, G=spruce

independently of location. MaxWP stands for a landscape management program that aims at high wood production with no other objective. MaxWP uses the stand management program "Wood production", which uses silvicultural measures that are permitted by the Swedish Forestry Act. For that reason digging of new ditches, clones and fertilising is excluded. MinN stands for a landscape management program that minimises the leaching of nitrogen to the ground water through use of stand management program "Water quality". BioCon stands for a landscape management program that aims at preserving biodiversity. The nature conservation measures being concentrated to areas inhabited by red-listed species, or areas in which the tree cover features can serve as habitat for red-listed species, so-called critical elements. BioDisp stands for a landscape management program that aims at preserving biodiversity by dispersing nature conservation measures. In BioDisp nature conservation and wood production are integrated in the majority of the stands. In these stands, management aims at creating mixed conifer and broadleaf forests in which a number of old-growth trees are left at the final felling. In a minority of stands, with characteristics close to the key habitat, nature conservation is the sole aim. The number of old-growth trees in BioDisp for the stand management programs WP-Spruce, WP-Pine and WP-Mixedconifer are birch 12 stems ha⁻¹, oak 11 stems ha⁻¹ and pine 15 stems ha⁻¹). In BioDisp for stand management program WP-Birch Long Rotation 12 birches are left as green tree retention per ha. The corresponding number for BioCon is 25 stems ha⁻¹.

13.6 Evaluation of the management programs

The outcome from the modelling is registered in indicators or criteria comparable with indicators or criteria expressing/quantifying objectives. It is evident that the programs have been effective

in the sense that each of the landscape management programs is "best" when evaluated in terms of the main aspect it was designed for. Thus, the Bio programs provide the best results in terms of the biodiversity relevant variables, while the MaxWP program yields the highest total return for the planning period and the MinN program is best in preventing the leaching of nitrogen. It is evident from the aggregated results for the different landscape management programs that no program is dominant in the sense of its being better than the others in all respects. They can be seen as providing initial inputs into what can become the future optimizing scheme for iteratively finding the maximum goal achievement. It should be noted that the outcome in biodiversity terms could not be describe in one-dimensional quantitative terms., BioCon, however, was judged to be considerably better than BioDisp. Thus, there is considerably potential for controlling the outcome in terms of biodiversity by employing different landscape management programs. The main conclusion that can be drawn is that it is possible to form very different landscapes over a longer time period with the management options available.

13.6.1 Effects on biodiversity on land

The landscape management programs MaxWP and MinN will probably eliminate many species and prevent colonisation of previously locally extinct species since these landscape management programs will reduce broadleaves, the numbers of trees of large diameter and the amount of dead wood. Many common species would also disappear from dense spruce stands (Magura et al. 2000). Thus, there is no reason to discuss these landscape management programs further from a biodiversity preservation perspective. It is important to highlight some aspects of the models used when interpreting the results of simulations. When modelling the diameter development, the calculations are based on the mean diameter of the trees in the stand. In reality, all stands consist of trees of different diameter classes, usually following a normal distribution. The diameter distribution depend, on stand history, management history/regimes (especially thinning programs) and the tree species involved. This means that some trees in a stand reach large diameters (40 cm, 70 cm) much earlier than the mean tree of the stand. Other deciduous trees than beech, oak and birch were not accounted for in the projections due to a lack of data and models describing the development of these other deciduous species. This is important when discussing biodiversity, since trees such as aspen, *Populus tremula* and willow *Salix caprea* may be of great importance for biodiversity. In the simulations, these species disappeared almost completely over time. However, in reality, if they had been favoured by silvicultural measures, they would instead have increased to above the present levels, especially in burned areas. The natural mortality patterns of trees and the development of important features that occur with higher tree age is largely unstudied in European tree species, due to the focus on managing forests for timber production. In such forests, rotation periods are about half or even less than the natural life span of the trees. The outcome of the simulation must thus be regarded as imprecise in terms of the absolute values and numbers of trees entering the diameter classes of 40-70 cm and >70 cm. We believe, however, that the general trends in mortality and death should reflect the real situation. It takes a long time, starting with the current situation, before a greater density of large dead trees is attained. Regarding large living trees, the density of dead trees >40 cm increased over the entire simulation horizon. On the other hand, the total density of dead trees of the dimension of 20-40 cm DBH was found to decrease over time. We have suggested that about 6 dead trees per ha with a DBH greater than 40 cm may be needed for biodiversity preservation and restoration in boreo-nemoral forests (Nilsson et al. 2000). After 240 years almost 5 such trees were attained, not far off target. In some stands, however, the densities was about 60 dead trees >40 cm per ha after 240 years, which seems very high. We may possibly have overestimated the decomposition times for some or all of the categories of dead trees. This underlines the need for studies of decomposition times for the largest dimensions (>40 and >70 cm) of the different tree species. The recently burned area is quite stable over time and amounts to approximately 2.3 ha yr⁻¹. One important result obtained for BioCon was the large number of large birches and pines that developed in the burned area. When

the area is burned again, these trees will be large enough to survive in large numbers and will provide very valuable habitats for threatened species. Furthermore, when these large trees die, they will be even more valuable since the threatened beetles species associated with dead pines and birches are considered to be favoured by sun-exposure (Gårdenfors and Baranowski 1992, Ehnström 1999). Quantitative studies appear to be lacking, however. The species dependent on burnt forests are mainly dependent on burnt trees and the ground below burnt trees, and not on burnt ground per se (Wikars 1997). Thus, the burnt patches in BioDisp are much less valuable than those in BioCon. This difference is probably important for species dependent on a low nitrogen concentration in the soil and a more or less constant open forest. We predict that dwarf shrubs e.g. *Vaccinium myrtillus*, *V. vitis-idaea* and leguminous plants would be more frequent in regularly burnt patches, provided the severity of each burning remain low or varies and providing grazing by deer is held under control to a greater extent than it is today. If so, regular burning would favour such species as *Capercallie Tetrao urugallos* and many butterfly species that have decreased continuously during the last 100 years. We conclude that the burning of forest land per se is of questionable value from a biodiversity point of view, whereas a well-developed strategy for burning may provide habitats supporting many threatened species. Overall, the tree species composition in the production layer, the shelter-wood layer and the old-growth layer changes very slowly, although after 180 years or more the proportion of spruce had decreased to about 50% of the basal area, whereas pine had increased to about 35%. Birches had increased to about 10% after 120 years. The differences in basal areas between BioCon and BioDisp are small for these tree species. On the other hand, the proportion of oak was twice as high in BioCon as in BioDisp after 180 years, due to the strong efforts to increase this species in area in which the stand management program Bio-Oak was employed. The tree species composition in the burnt area in BioCon converged to nearly equal proportions of pine and birch with no other tree species. However, if favoured by management measures, aspen, willow, oak and spruce ought to be found in the burnt areas. In a burnt forest in southeastern Sweden the numbers of seedlings per m² one year after the fire were: 90 pines, 60 aspens, 30 spruces, 20 birches and 7 willows (Granström 1991). The model for succession on burnt and managed forest land could be developed further, since the special characteristics that burning causes affect the results of regeneration. Both the lesser spotted woodpecker, *Dendrocopos minor*, and the white-backed woodpecker, *Dendrocopos leucotos*, require at least 20% older deciduous trees within an area of at most about 200 ha in order to survive (Nilsson et al. 2000). The two patches in BioCon with old deciduous trees and burnt forest, respectively, provide suitable habitats for at least two pairs of the regionally extinct white-backed woodpecker, and for the other woodpeckers as well. It is questionable, however, whether BioDisp can provide suitable habitats for the white-backed woodpecker, due to the fragmented pattern of deciduous trees.

13.6.2 Nutrient budgets

All landscape management programs remove nutrients from the soil, including base cations. Although, the harvest is dominated by stem wood, the branches also represent a substantial loss of base cations due to their relatively high content of nutrients, despite the calculations assuming no harvest of needles. Branches are not harvested in MinN. MaxWP shows the largest harvest in terms of biomass. Soil weathering represents an important input of base cations to forest soils. To simplify, the mass balance calculations of base cations include three levels of weathering rates were employed for all stands. The mass balance calculations of base cations comprise the atmospheric deposition and the approximate weathering minus biomass removal by harvest and leaching. The input data to the mass balance calculations are pertain to individual stands, except for weathering. The results obtain, presented as annual average data for the four management programs, are shown in Table 13.11. The mass balance of the sum of base cations (BC) is calculated for each of the three weathering rates separately. A negative mass balance means an average net loss of base cations across the simulation horizon. The variation of the mass balance between the 2,500 forest stands in the study area is large,

| Landscape management program | Stem wood | Branches | Needles |
|------------------------------|-----------|----------|---------|
| MaxWP | 2.53 | 0.20 | 0 |
| BioDisp | 2.12 | 0.11 | 0 |
| BioCon | 2.18 | 0.15 | 0 |
| MinN | 2.42 | 0.00 | 0 |

Table 13.10: *Biomass removal by harvest from 2,870 ha, average ton ha⁻¹yr⁻¹ during 240 years*

due mainly to tree species, production and harvest intensity. The percentage of the area with net loss of base cations (median weathering for all stands), as calculated for each landscape management program, is also presented. Table 13.11 shows the net loss of Ca and Mg for each

| Landscape management program | Ca | Mg | K | Sum BC W Median | Negative BC % of area | Sum BC W low | Sum BC W high |
|------------------------------|--------|--------|-------|-----------------------|--------------------------|--------------------|---------------------|
| MaxWP | -0.084 | -0.017 | 0.033 | -0.068 | 55 | -0.173 | 0.010 |
| BioDisp | -0.063 | -0.008 | 0.042 | -0.029 | 42 | -0.135 | 0.048 |
| BioCon | -0.060 | -0.009 | 0.041 | -0.028 | 46 | -0.134 | 0.049 |
| MinN | -0.029 | -0.002 | 0.052 | 0.021 | 35 | -0.085 | 0.098 |

Table 13.11: *Mass balance of base cations in forest soil (2,870 ha), average in keq ha⁻¹yr⁻¹ over the simulation horizon. Calculations of individual elements have used median weathering in the mass balance (BC = Base cations)*

of the landscape management programs. The net loss is highest for the MaxWP program due mainly to the greater removal of biomass and nutrients. The MinN program shows the lowest net loss, mainly because of there being no harvest of branches. K shows a positive balance in all the programs, due mainly to the relatively high weathering rate of K in the Asa area. The sum of the base cations shows a net loss from the soil. The present atmospheric deposition of acidifying air pollutants is low compared with the period of 1960 to 1990. The high deposition, and to some extent the high production and uptake of base cations in the trees, have decreased the base saturation in the forest soils of large parts of Sweden. The model calculations reveal an average loss of 50% of the base saturation down to a depth of 50 cm in forest plots in southern Sweden (Moldan et al. 1999). The average loss of base cations in the soil during the last 100 years of that study was estimated to be approximately 26 keq ha⁻¹. This can be compared with the calculated losses in the case study during a single rotation (80 years) being 2.2 keq ha⁻¹ (BioCon), 2.3 keq ha⁻¹ (BioDisp) and 5.4 keq ha⁻¹ (MaxWP), respectively.

The comparison shows the average losses of base cations in the different management programs to be moderate (10% to 30% of total losses 1970-2000) as compared with the effects caused by the high historic deposition of acidifying air pollutants. To this it can be added that the weathering rate is the property of the forest owner and is his asset to be used as raw material in forest production, whereas the use of weathering for the neutralization of trans-boundary pollution is an act of trespassing on property and represents the unsolicited use of it. This is aggravated by the possibility that use of the weathering rate can be strongly affected by this trespassing, making it of paramount importance to either strongly reduce acid deposition to a level of full compliance with critical loads, or to create an international invoicing

| Program | Number of large living/dead trees, ha ⁻¹ | Biodiversity indicators | | |
|---------|---|--|--|--|
| | | The increase of the number of large living trees | The increase of the number of large dead trees | Spatial distribution of burnt trees and old living trees |
| BioCon | 0.7/4.7 | Faster | Slower | Concentrated |
| BioDisp | 1.05/4.5 | Slower | Faster | Fragmented |

Table 13.12: *A compilation of predicted outcomes obtained from the forest management programs BioCon and BioDisp.*

scheme through which the polluters are made to pay the full and complete costs of what they are presently doing through trespassing and violation property rights both to income and stock, both visible and hidden. The consequences of the net loss of base cations is a risk for long-run nutrient imbalance. The calculations assume the same nutrient uptake by trees and the same leaching during the entire period independent of the net changes in the soil. The depletion of base cations can be expected to decrease the concentrations of them in surface water, which can enhance the existing problems regarding acidified streams. Historically, stream water provided lower concentrations of base cations than at present, present leaves either being elevated because of acidification and the leaching of base cations, or decreasing because of far-reaching soil depletion. The effects of soil acidification and of aluminium on nutrient uptake and tree vitality have been systematically ignored in the four scenarios. Any direct effect of air pollution on tree growth and vitality have also been ignored in the present assessments. Two landscape management programs, MaxWP and BioDisp, were used for evaluation of the probable risks to forest health at the end of the simulation horizon. It is evident from the base cation budgets that MaxWP was not sustainable in large parts of the case study area, due to an obvious risk of nutrient imbalance in the trees. In MaxWP, in which there is a dominance of Norway spruce, 17% of the area was found to have a negative budget for the important base cations, Ca, Mg and K, whereas in BioDisp the area with a negative Ca, Mg and K budget was 13%. With a higher percentage of deciduous tree species and a longer simulation horizon, the area in which there is a negative nutrient budget may have decreased still further. In BioDisp, Scots pine, Norway spruce and birch are found in mixtures to a greater extent. Even this landscape management program, however, showed negative values for the Ca and Mg budgets, there was no difference in risk assessment between the two landscape management programs, although with another mixture of tree species the result might have been different. In MaxWP, Norway spruce is the dominant tree species. A negative Ca budget further increases the risk of attacks by pests and pathogens. A decreased Ca availability predisposes the tree to diseases that weaken or penetrate the cell walls (McLaughlin and Wimmer 1999). On the sites with a negative K budget, the risk of Armillaria infection increases (Moore et al 1993).

13.6.3 Effects on water quality

The most important factor affecting the leaching of nitrogen is the annual area that is clear-cut. The area clear-cut or regenerated under shelter-wood varies according to the landscape management program. The highest leaching occurs in the MaxWP program. The leaching over time follows approximately the presence of clear-cut areas, the highest values being expected to occur 50-100 years from now. The mean difference between the MaxWP and the MinN program is approximately 0.16 kg N ha⁻¹yr⁻¹. The reason for the rather small difference between the programs is that in the MaxWP program large areas are also regenerated under shelter-wood. If instead, all areas had been clear-cut in the MaxWP-program the difference in leaching as

compared with MinN would have increased by a factor of 3 (from $0.16 \text{ kg N ha}^{-1}\text{yr}^{-1}$ to ca $0.48 \text{ kg N ha}^{-1}\text{yr}^{-1}$). The difference between the two programs was rather small due to the fact that we have calculated the same base leaching ($2 \text{ kg ha}^{-1}\text{yr}^{-1}$), and that leaching only peak during the relatively seldom occurring clear-cuttings.

13.6.4 Effects on tree vitality

In the following, the outcomes of the landscape management programs MaxWP and MinN in terms of frost and wind damages are assessed from a wood production perspective. Of the different kinds of frost damage that may occur, radiative frosts during the growing season mainly damage short vegetation growing in locations well exposed to the cold night sky, such as tree seedlings on a clear felling. The frost-risk model is one step towards developing a tool for assessing the risk of frost damage of this type, one that can be applied to a landscape, such as to the present case-study area. The next step will be to couple this model of relative risk with a model for simulating the effects of shelter-wood and of scarification (Blennow 1998), two silvicultural measures for reducing the risk of exposure to low temperature during the growing season. Furthermore, the frequency of low temperature events during the growing season and the cold hardiness of the seedlings need to be considered. The distribution of frost-hazard classes for clear-felled areas within the case-study area is not expected to change unless the relative importance of the different processes gives rise to changes in temperature pattern. Such changes can be expected after marked alterations, such as the extensive clearing of a forest. This is not a characteristic of any of the different landscape management programs evaluated in this study. The model can thus be used for determining the distribution of the relative risk of frost occurrence on cleared areas during the growing season. When discussing the relative risk of frost damage, the model will be less useful as the number of tree species increase, since species react differently to low temperatures. The following discussion will focus on the two landscape management programs with few tree species involved, Maximum Wood Production (MaxWP) and Minimum Nitrogen Leaching (MinN). Large proportions of area under regeneration will make the landscape less robust in a frost damage perspective, at least as long as no risk reducing measures are taken. Such measures could involve the use of shelter-wood, scarification, frost-hardy provenances or species for planting and consideration of management program in relation to frost hazard class. Except for the choice of a frost-hardy plant material, these measures are included in the landscape management programs Maximum Wood Production (MaxWP) and Minimum Nitrogen Leaching (MinN). Comparing the two programs, in MinN a shelter-wood is always left at final felling whereas in MaxWP shelter-wood is used at sites of medium or low fertility and at sites classified as being frost-prone. In the simulations, the acreage of young forest which is not growing within shelter-wood (growing exposed to the cold night sky) is slightly larger for the MinN program than for the MaxWP program (10-15%). In discriminating in terms of frost hazard class between different locations in the landscape, the acreage of exposed young forest in the highest frost hazard class is larger for the MinN (0-15%) than for the MaxWP program. This is probably an effect of the timing of shelter-wood removal, which is carried out 5-9 years after the final felling, in which case the difference in acreage does not primarily apply to the early part of the young forest phase, which is more important for the risk of frost than the latter part of the phase is. For the young forest phase as a whole, MinN appears somewhat less robust in terms of frost hazard than MaxWP. However, the frost risk reducing measures employed in both programs should effectively reduce the frequency of frost events when the seedlings are less than 1 m in height. In comparing the two programs in terms of risk of frost damage, the resistance of the tree species to low temperature needs also to be considered. In the landscape management programs, Norway spruce and Scots pine are favoured over other species. Seedlings of Scots pine are more resistant to low temperature than Norway spruce are (Christersson and von Fircks, 1988). In the simulations, Scots pine increased in number over time compared to Norway spruce, especially in the case of the MinN program. Regarding the risk of frost damage, this difference in species composition is probably more important than the

contrasting effect between the landscape management programs in terms of frost hazard.

In order to be able to evaluate the landscape management programs MaxWP and MinN, parts of the system of models were run for each program at four points in time during the simulation period. At each point in time and for each program, the number of stands having at least one 10 m edge was calculated. The number of stands with such edges was initially larger for the MinN program. After 180 years of the simulation horizon, however, the conditions were reversed, this indicates the risk of windthrow to be larger for MinN than for MaxWP until 180 years from present. Considering the larger amount of shelter-wood in the MinN program, as described above, and of the fact that shelter-woods are less resistant to strong winds, it is likely that the risk of windthrow, when considered for the simulation period as a whole, is greater for the MinN than for the MaxWP program. However, it is not only the number of stands with 10 m edges in a landscape that determines the risk of windthrow but also the variation in wind direction. This means that the direction of an edge is of importance. Furthermore, the distribution of wind speeds over the landscape varies with direction of the wind and with land-use. Thus, the frequency of direction of strong winds and the location of a stand edge in the terrain are important as well. This means too that it is possible to influence the risk of windthrow by taking the location in space, preferably at a landscape level into account, when choosing management measures. Additional ways of affecting the risk of windthrow include choice of silvicultural treatments, such as thinning regime (Nielsen 1995) and the number of canopy layers within a stand (Gardiner 1995). The latter can be expected to increase when applying stand management programs aimed at preserving biodiversity, in which case the risk of windthrow is reduced. Another factor affecting the resistance to windthrow is the tree species composition. Since in southern Sweden strong winds are more frequent during the period when deciduous trees have shed their leaves, introducing a greater number of deciduous trees into a landscape can be expected to reduced risk of windthrow. In assessing the risk of windthrow several of the aspects mentioned above will be evaluated quantitatively using the system of models presented.

13.6.5 Economic effects

The growing stock varies over time between 120 and 230 m³sk ha⁻¹, depending on the age class distribution. For the landscape management programs MaxWP and MinN, the average levels during the first and last 90 years of the simulation are about equal whereas for the BioCon and BioDisp programs the growing stock is higher during the first period than during the last. In all the landscape management programs, the tree species composition changes over time. In the MaxWP and MinN programs, the proportion of broad-leaves decrease towards 0 during the first 100 years and remain there. The proportion of spruce also decrease during the first 150-200 years of the simulation, there in contrast being an increase in the proportion of pine during that period. For the BioCon and BioDisp programs, the changes in tree species composition are towards more broadleaves and pine, and less spruce. In BioCon, the proportion of broad-leaves rises from 5% to 15% by 180 years from the present point in time, which is the maximum proportion during the simulation. The corresponding figures for BioDisp being 6% to 18%. For BioCon and BioDisp, the changes in broad-leaves proportions over time show a similar pattern, the increase in the proportions of pine and the decrease in the proportions of spruce being similar in magnitude and pattern over time for those two programs. In general, the changes in the proportions of pine and spruce occur during the first 90 years, the proportions remaining relatively stable during the last 150 years of the simulation. In all landscape management programs, applied management methods have full effect on tree species composition 150 years from present. The diameter distributions varied little between the different landscape management programs. Over time, the management programs resulted in an average wood removal of around 7.0 m³sk ha⁻¹yr⁻¹, whereby for MaxWP it was 7.2 m³sk ha⁻¹yr⁻¹, for MinN 6.9 m³sk ha⁻¹yr⁻¹, for BioDisp 6.5 m³sk ha⁻¹yr⁻¹ and for BioCon 6.5 m³sk ha⁻¹yr⁻¹. The reduction in wood removal is 4% for MinN and 9 and 10%, respectively, for BioDisp and BioCon. The differences are due to the

| Program | N leaching compared with MaxWP | Net return compared with MaxWP | Reduction in net return for reducing N leaching one kg |
|---------|--|---------------------------------------|--|
| | kg N ha ⁻¹ yr ⁻¹ | SEK ha ⁻¹ yr ⁻¹ | SEK kg(N) ⁻¹ |
| MaxWP | 2.19 | 1,539 | - |
| BioDisp | 2.15 | 1,387 | 3,779 |
| BioCon | 2.15 | 1,369 | 3,771 |
| MinN | 2.03 | 1,445 | 587 |

Table 13.13: *Reduction in N leaching and net return compared with MaxWP. Reduction of net return for reducing N leaching by one kg in the landscape management programs MinN, BioCon and BioDisp calculated as reduction in net return divided by reduction in N leaching*

current age class distribution and to the wood removal peaks 90 years from the present as a result of a large proportion of the forest being ready for final felling. The decrease in wood removal over the simulation horizon is connected to the lower growth caused by the transformation of the present Spruce forests to mixed Spruce and Pine forests. For all the landscape management programs, removals are lesser in magnitude than growth, seen over the simulation horizon as a whole. The proportion of timber in the total wood removed varies with the age class distribution but remains for all landscape management programs at 30% on the average during the simulation horizon. The microeconomic analyses carried out in this study focused on the costs and incomes that originate from silviculture, logging and wood sale. The microeconomic output is calculated by subtracting the costs that originate from the goods and services needed for silvicultural measures and the logging activities from the incomes that originate from sale of wood. Other costs, such as for road building, planning, administration, or incomes, such as incomes from hunting, are not included. Over time the relative net return decreases for the management programs MinN, BioDisp and BioCon as compared with MaxWP. The differences are a few percent at the beginning of the simulation horizon and increase to 15-20% at the end. This pattern is due to the present forest conditions of high growth of valuable wood being exploited in all programs during an early part of the simulation horizon and its only being fully restored in the MaxWP program. Late in the simulation horizon the BioDisp and BioCon programs can no longer benefit from the present forest conditions and the relative net return decrease, since the forest conditions change as a result of management, the differences in net return increase.

13.7 Conclusions

Different strategic landscape management programs were formulated differing in their objectives and effective in achieving the outcomes aimed at. The degrees of freedom in the creation of different scenarios for future landscapes is large, and a considerable freedom of choice exists. The cases studied suggest the order of magnitude for the costs required for meeting multiple goals. Based on the assumptions made, the results were as follows: The costs of biodiversity, in terms of the reduction in the average net return over the simulation horizon (240 years) were 10%, less at in the beginning of the simulation horizon (0 to 5%) than at the end (15 to 20%). The benefits of biodiversity increased over time as the economic return dropped. Nothing was gained economically by concentrating nature conservation. Various ways of improving the landscape management program BioCon are suggested in order to point out the potential a concentrated strategy can possess. The leaching of nitrogen is caused mainly by the high

| Program | Ca | Mg | Dolomitic limestone | Wood ash | Fertiliser Skogvital |
|---------------------|------|-----|------------------------|-------------|-------------------------|
| Kg ha ⁻¹ | | | | | |
| MaxWP | -135 | -17 | 620 | 900 | 1,020 |
| BioDisp | -101 | -8 | 470 | 670 | 760 |
| BioCon | -96 | -9 | 440 | 640 | 722 |
| MinN | -47 | -2 | 220 | 310 | 350 |

Table 13.14: *Approximate losses of base cations and approximate doses of fertilisers for compensation during one rotation (80 year), kg ha⁻¹. Doses of wood ash are based on average nutrient concentrations after combustion of branches and tops. The utilization efficiency of the product and the purity of the products vary a lot, efficiency from as low as 30% for coarse dolomitic crush to nearly 100% in fine-grained ashes, the purity vary from 75% to 95% depending on origin.*

| Item | Maximum production | Minimum N leaching | Biodiversity concentrated | Biodiversity dispersed |
|--|-----------------------|-----------------------|------------------------------|---------------------------|
| Growth average (m ³ sk ha ⁻¹ yr ⁻¹) | 7.6 | 7.2 | 7.0 | 7.1 |
| Growth end of simulation (m ³ sk ha ⁻¹ yr ⁻¹) | 7.4 | 7.0 | 6.9 | 6.8 |
| Removal average (m ³ sk ha ⁻¹ yr ⁻¹) | 7.2 | 6.9 | 6.5 | 6.5 |
| Net return (SEK ha ⁻¹ yr ⁻¹) | 1,539 | 1,445 | 1,369 | 1,387 |
| Area with BC deficit | 55% | 35% | 46% | 42% |
| Productive loss in stock (m ³ sk) | 6 | 8 | 20 | 20 |
| Leaching N (kg ha ⁻¹ yr ⁻¹) | 2.19 | 2.03 | 2.15 | 2.15 |
| Fertilizing need (t ha ⁻¹) | 1-3 | 0.7-2 | 0.7-2 | 0.3-1 |
| Risk summary | | | | |
| Soil acidification stress | (high) | (high) | (partial) | (moderate) |
| Air pollution stress | n.a. | n.a. | n.a. | n.a. |
| Nutrient deficiency stress | high | high | partial | moderate |
| Frost damage | moderate | less | n.a. | n.a. |
| Wind damage | moderate | high | n.a. | n.a. |
| Browsing | 1.94 | 2.07 | 2.68 | 2.87 |
| Pine weevil | 78.4 | 56.5 | 60.1 | 62.2 |
| Bark boring insects | 30 | 29 | 26 | 25 |
| Butt rot | high | moderate | less | less |
| Time to one oak > 70 cm BRH) per 20 ha (yr) | n.a. | n.a. | 120 | 180 |
| Spatial pattern of large living and dead trees | n.a. | n.a. | Concentrated | Dispersed |
| Biodiversity evaluation | n.a. | n.a. | ++ | + |

Table 13.15: *Summary of assessment results*

deposition and is partly outside the control of forestry. However, the leaching of nitrogen can be reduced only slightly, by 10%, over a long period of time, when the amount of final felling is high the reduction can occasionally be as high as 20% its being due in this case to shelter-wood being left at final felling. The effect on nitrogen leaching of slash removal and scarification was, based on empirical evidence, assumed to be smaller than that of leaving dense shelter-wood ($12 \text{ m}^2\text{ha}^{-1}$) instead of clear-cutting. The marginal costs of the reduction in nitrogen leaching were 587 SEK kg^{-1} . In comparison with other ways of reducing nitrogen leaching that are not related to forestry, use of shelter-wood appeared to be an expensive measure. The nutrient budgets for base cations, Ca, Mg and K, calculated across the simulation horizon and for the entire landscape, were negative for 30% to 50% of the area, depending on the landscape management program involved. Ca was most negative of all, followed by Mg. Although individual stands showed both negative and positive budgets for Ca, Mg and K, it appeared that the costs for base cation replacement were not very large. It is important to note, however, that even if forestry is presently paying for the costs of pollution in terms of damage to the base cation resources. The risk of a negative outcome in economic terms is important to consider in evaluating landscape management programs. A landscape management program can produce a high return but be associated with a high level of risk. Considering both the return and the risks involved, the landscape management program MinN appears to be of considerable interest, since it reduces such risks as those for frost damages, pine weevil, bark-boring insects and but rot, and only reduces return by 6%, as compared with MaxWP. However, the risks of wind damages and browsing are higher for MinN. Both the returns and the risks need to be taken into account in order to adequately evaluate the economic outcome of a landscape management program.

Chapter 14

Assessment of sustainability in the Asa Forest Park

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14.1 Assessment of the sustainability of mineral nutrient use

14.1.1 Introduction

The main objectives of the geochemical investigation in Asa Forest Research Park (referred to henceforth simply as Asa) were as follows:

1. to calculate the release of Ca, Mg, K, and P from mineral weathering using PROFILE, a biogeochemical soil model,
2. to determine the geographical distribution of the results of weathering rate calculations carried out both at the site level and on a regional scale and
3. to propose a mass balance method for calculating the optimal sustainable biomass production of a forested area in southern Sweden

The weathering rate is an important element in assessing and understanding the principles behind nutrient sustainability in forest management. It is also important to an understanding of how different soil types react to various anthropogenic influences, like acid deposition and forest management. The geochemical study at Asa involved primarily sampling and analyzing the forest soil to determine the soil type and other chemical and physical soil parameters. The sampling strategy was to collect sufficient data to enable a weathering rate map of Asa to be produced. The sampling technique chosen was to carry out a geostatistical analysis in order to be able to construct by, means of kriging, an interpolated map of the weathering rate and the sustainable biomass production.

14.1.2 Data collection

The data for the climatic parameters that the model employs were collected at the Asa Research Station (Langwall 1999). The same applies to the data on stand characteristics. The wet and dry atmospheric deposition data for anions and cations used, were taken from the Swedish national critical loads database, derived originally from deposition estimates made by the Swedish Environmental Research Institute, IVL (Hallgren-Larsson 1999). All other data used in the modelling are default values or data collected in the geochemical investigation. In spatial surveying of all kinds describing and analyzing variations in phenomena, both natural and man-made,

over the land surface is important. Although, many geographical and geochemical properties vary continuously and randomly in space, the pattern and scale of their variation is often not readily apparent. These include rainfall, air and ground temperatures, atmospheric deposition, particle size distribution, soil chemistry and the occurrence of different minerals in the soil.

An underlying problem in two-dimensional mapping of all kinds is to distribute a point value obtained to the area surrounding it. To accomplish this it is necessary to sample in such a way that one has the possibility of estimating the spatial uncertainty. Therefore, one needs to use a sampling density that is sufficiently high but is also manageable in both an economic and a practical way and to use a sampling strategy that is relevant for the geochemical investigation in question. It is also important to consider what degree of precision the investigation is aimed at. The attempt to achieve high spatial precision and low spatial uncertainty calls for a high sampling density.

14.1.3 Soil sampling

The soil sampling in Asa Research Park was performed in autumn of 1998 and during June to November of 1999. 89 soil pits were excavated in 1998 and 273 more soil pits were dug in 1999, a total 362 excavations altogether (Figure 14.1). A separate soil sampling strategy was used for each of the two sampling seasons. In 1998, a logarithmic distribution pattern was employed, involving high sampling density in

the center and a more even and dispersed density in the peripheral areas (Figure 14.1 and 14.2).

For the soil sampling conducted in 1999, a nested survey technique Webster et al. 1977, Oliver et al. 1986, Oliver et al. 1987) was used. This technique represents an adaptation of multi-stage hierarchical sampling in which the stages of the design represent particular sampling intervals. The nested sampling involves a number of soil sampling centers or “spiders” spread out randomly over the area investigated. In this study there were eight such centre-locations for soil sampling distributed over the area of the park. These centres were located at distances of 600-800 meters from each other. At the next stage (2) additional sampling sites were selected, each of them located approximately 180 meters in a randomly selected direction from the center point in question, 8 new soil pits being dug in the area around each center. At stage (3), sampling sites were chosen approximately 80 meters from each of the points determined at stage (2), 16 new soil pits in each case. At stage (4) sampling sites were chosen approximately 30 meters

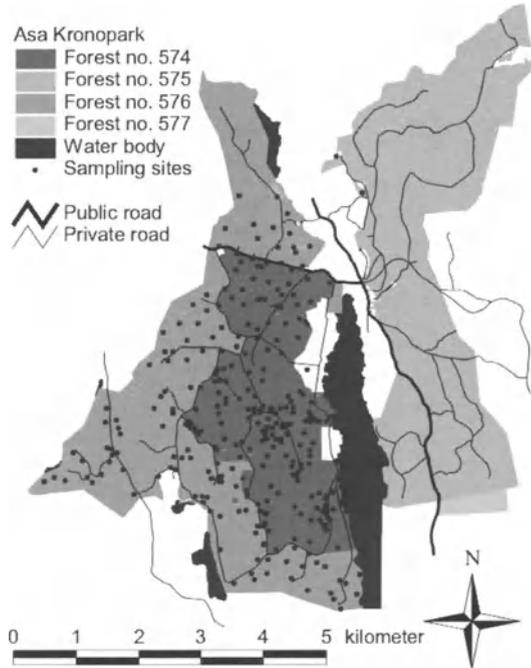


Figure 14.1: Map over the distribution of the soil pits that were excavated 1998-1999. The map shows the Asa Royal Forest Park including the Asa Research Park. The traditional forest yield is shown as distributed among the registered forest stands at Asa, units are $m^3 \text{ stemwood ha}^{-1} \text{ yr}^{-1}$. In the first part of this programme, only the part west of the Lake Asa was mapped.

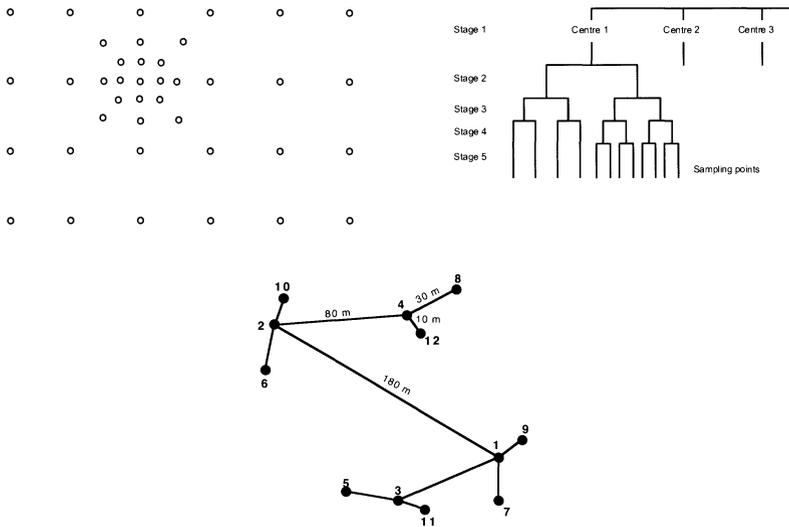


Figure 14.2: A schematic presentation of the principles basic to the sampling strategy used in the field during the 1998 season. The spatial configuration of a set of sampling points, a so-called "spider", illustrates the principle of nested sampling. At a given stage each point has a fixed distance to the point it is related to, but the direction to it is selected randomly.

from each of the points determined at stage (3) and finally at stage (5), sampling sites were chosen approximately 10 meters from each of the points determined at stage (2), see Figure 14.2 and Table 14.1. Altogether, 96 soil pits were included in this nested sampling. To cover the entire Asa Kronopark area west of Asa Lake (Forest nos. 574, 576 and 577 in Figure 14.1), an even sampling net with a sampling distance of 300 m was also spread over that area. A third sampling strategy was also adopted in the sampling season of 1999.

Use of these sampling strategies allows the sampling density to be optimized. The nested sampling technique enables one to estimate the variance there is for a defined sampling density with use of different measured and analyzed parameters and vice versa. This makes it possible to perform a cost-benefit analysis of the geochemical soil study. Such a geostatistical study can also be of considerable help in mapping other areas with similar intent. In addition, one can assess variations in the uncertainty of the distributed data within in the area, as a function of the soil sampling density.

14.1.4 Calculation flowchart

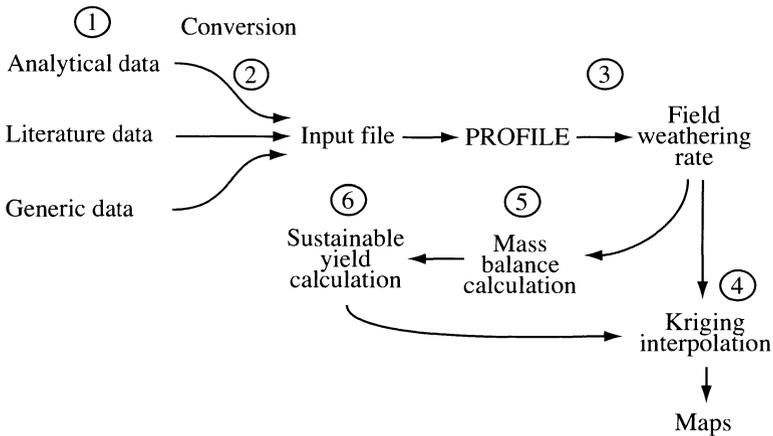
The strategy that was used to calculate nutrient mass balance and sustainable yield in forest ecosystems is shown in Figure 14.3. The assessment to estimate the sustainable yield could be summarized by the steps:

1. Gather information using analytical results from the geochemical study in Asa, such as determination of the elemental composition and grain-size analysis, generic data from other sources on factors such as temperature (Asa Forest Station) and deposition (IVL, Swedish Environmental Research Institute) and also for example root depth and efficiency
2. Convert the data-sets to a readable input data file for the PROFILE model.

| Stage | Sampling interval (m) | Number of replicates |
|-------|-----------------------|----------------------|
| 1 | 600 | 8 |
| 2 | 180 | 16 |
| 3 | 80 | 32 |
| 4 | 30 | 64 |
| 5 | 10 | 96 |

Table 14.1: *Sampling intervals and replication in the nested sampling scheme.*

3. Calculate the weathering rate for calcium, potassium and magnesium by the PROFILE model
4. Determine site specific weathering rate where spatially distributed by kriging interpolation
5. Estimate weathering rate, deposition, uptake and leaching for use in calculating the mass balance
6. On the basis of the mass balance calculation, estimate the sustainable yield in Asa.

Figure 14.3: *Schematic flowchart showing the principal steps in estimating the sustainable yield in Asa, a forested area in southern Sweden.*

14.1.5 Measured soil parameters and analytical methods

A major aim of the geochemical study of the forested area in Asa was to collect sufficient data to model the area in terms of weathering rate and growth rate. Thus, strong efforts were made to characterize the soil physically and chemically. The exact point at which soil was to be sampled was selected in terms of its being sufficiently easy with respect to tree-roots, stoniness, vegetation, and topography. A spot was also selected to represent an area of the stand with a normal to high growth rate, unless the position had been decided on beforehand. The soil pits were about 40 cm in diameter and were preferably dug down to the unaltered C-horizon, provided this was not deeper than 100 cm below the surface. If the B/C-horizon continued on down below

100 cm the deepest soil sample was taken at 100 cm. Two horizons were sampled in each soil pit. The soil samples were taken from the upper B-horizon and the C-horizon the direction in which the material was collected in the B- and the C-horizon of the soil-pit differing to obtain as representative a soil sample as possible. The sampling technique used for the O/H-horizon depended on the spatial heterogeneity of these two uppermost soil horizons. In a square, 10 m in length on each side, the pit being in the centre, 8 samples were collected with a humus-auger and homogenized in order to obtain a single representative soil sample from the O/H-horizon. At each location, records of the depth and colour of the different horizons, the soil type and the slope angle were made. Vegetation data were also recorded. An estimate of the tree growth-rate near the location was also made on the basis of data obtained from a full-grown tree using an elevation-meter and a tree ring width estimator. The physical soil properties measured for each of the soil samples were texture, soil moisture and bulk density. The texture was determined by a conventional grain-size analysis which involved sieving the sand and gravel fractions using a sieve interval of 0.5 mm. Fractions finer than 2 mm were analyzed by means of the hydrometer method. Surface area was also analyzed for 18 samples by use of the BET-technique, in order to compare the grain-size analysis with the surface area according to BET measurements. To estimate the surface area from results of the grain-size analysis an empirical algorithm was used (14.1, 14.2) one developed by Sverdrup (1990) and specified further in Warfvinge and Sverdrup (1995). Texture was only determined on soil-samples from the C-horizon.

$$A_W = (0 \cdot X_{coarse} + 0.3 \cdot X_{sand} + 2.2 \cdot X_{silt} + 8.0 \cdot X_{clay}) \cdot \rho_{bulk} \cdot 1000 \tag{14.1}$$

$$X_{coarse} + X_{sand} + X_{silt} + X_{clay} = 1 \tag{14.2}$$

The moisture content, *Moist (%)*, which is very much dependent on when the soil sample was collected, was determined by the difference between the dried and the moist soil sample. The bulk density was determined by weighing a known volume of dried (105°C. for 24 hours) soil.

The chemical soil properties analyzed for the different horizons were pH water and pH BaCl₂, carbon content, loss of ignition (LOI), nitrogen content, and the elemental composition of the solid particles following BaCl₂- and EDTA-extraction. The methodology employed was that used by the convention of the United Nations Economic Commission for Europe on Long-Range Trans boundary Air Pollution (1998) used in the "International Co-operative Programme on Assessment and Monitoring of Air Pollution Effects on Forests". The pH and exchangeable cations were determined by placing a 10 g humus or 20 g mineral soil sample in a 200 ml bottle, adding 100 ml of 0.1 M BaCl₂ solution, shaking the mixture for 2 hours, filtering it and analyzing the filtrate on ICP-AES (Perkin Elmer, Connecticut, USA) with regard to the base cations Ba, K, Ca, Mg, and Na together with acid cations such as Al, Fe, and Mn. The amount of hydrogen per gram soil was determined using equation 14.3.

$$H^+ (\mu eq g^{-1}) = \frac{-\log^{pH_{BaCl_2}} \cdot V_{0.1M BaCl_2}}{m_{fine\ earth}} \tag{14.3}$$

CEC was calculated by taking the sum of all exchangeable cations and the base saturation was calculated as (14.4):

$$Base\ saturation(\%) = \frac{Sum\ base\ cations}{CEC} \cdot 100 \tag{14.4}$$

Exchangeable metal ions, such as Cu, Zn, and P, were determined through extraction by an acid 0.02 M EDTA solution. The extraction of the metal ions followed the same procedure as for the extraction using BaCl₂. To determine the elemental composition of the minerals in the C-horizon, the soil-sample was preheated to 550°C for 2 h, 0.1 gram of the ash being mixed then with 0.5 g of the flux Li₂B₄O₇. The mixture was heated to 1000°C for 15 min. and fused to a homogeneous melt. The melt was dissolved in heated 5% HNO₃. The amount dissolved in the Li₂B₄O₇ melt was analyzed on ICP-AES with regard to the elements Si, Al, Fe, Ba, K,

Na, Ca, Mg, Mn, P, Zn, Zr and Ti. The Kjeldahl method was used to analyze N. The carbon content was determined by LECO, at 550°C and the soil organic carbon content was determined through loss on ignition LOI (Bengtsson 1986).

Mineralogical analyses

The mineralogical analyses used in the study were a SEM-EDX-technique and a new XRD-technique. Both techniques provide a quantitative estimation of the mineralogical composition. The analyses were performed on the silt fraction 30-63 μm of the C-horizon. Macaulay Research & Consultancy Services Ltd, of Aberdeen, Scotland, performed the XRD-analyses. The bulk samples were ground wet and were spray dried to produce a random powder to which 20 wt% corundum was added to act as an internal standard for quantitative phase analysis (QPA). QPA was carried out using a reference intensity ratio (RIR) method. The accuracy (?) was estimated to be better than ± 3 wt% at the 95% confidence level. The SEM-EDX analyses were performed at the Electron Microscopy Unit at the Faculty of Medicine, Lund University, using a Philips SEM 515 with LINK ISIS energy dispersive X-ray microanalyser (EDX). The sample analyzed by SEM-EDX was the same as that analyzed by XRD. The sample was coated with carbon to minimize interference with other elements. The elemental composition of 300 silt particles was determined for each sample. The spectra and the quantitative elemental composition of each particle were compared with standard minerals. This made it possible to determine the content of quartz, K-feldspar, albite, anorthite, biotite, apatite, zircon, titanite, and of a mineral group of dark minerals that included amphiboles, pyroxenes, epidote and garnet. It was also possible to determine the average elemental composition of each mineral and mineral group. Altogether 18 samples were analyzed by XRD and 10 samples by SEM-EDX.

Mineral normalization

Calculation of the mineralogical composition involved use of the elemental composition of the soil sample and of a simple normalization model. The UPPSALA model we used are based on the (mathematical) principle that the mineral composition can be expressed from a linear combination of the total element analysis data (Warfvinge and Sverdrup 1995). The UPPSALA model was modified slightly in this study to be more adequate to the mineralogical knowledge about the area that was available. The modified UPPSALA model used at Asa is shown in Tab. 14.3. Since normalization of the soil mineralogy at Asa yields no chlorite, the composition was set to yield approximately equal amounts of muscovite and biotite. The soils at Asa are relatively rich in K-feldspar and plagioclase. The weathering rates; as well as the rates of release of the ions of calcium, magnesium, potassium, and phosphorus; for the rooting zone were calculated by use of a regional version of the PROFILE model.

Soil texture and mineralogy surfaces

The area of the minerals surface in the soil involved in weathering was estimated using a texture classification system. This system can be converted into the system used in the Swedish Forest Inventory, namely

$$TX = 6 - TX_{Asa} \quad (14.5)$$

The classification can be determined on the basis either of the particle size distribution or of BET analysis.

$$TX = \frac{\ln A - \ln 0.093}{0.51} \quad (14.6)$$

where TX is a texture class according to the classification scheme of the National Swedish Forest Inventory. A given class can be converted to surface area in m².

$$A = 0.093 \cdot e^{0.51 \cdot TX} \quad (14.7)$$

The texture class was determined at each of the 362 Asa sites.

14.1.6 Results

Spatial geostatistical analysis

Describing and analyzing the spatial variations of phenomena over a land surface is of central importance in earth science and in other geographical disciplines. Soil scientists, forest ecologists, forest managers and others working with spatially distributed data have thus devoted considerable attention to defining regional patterns in natural landscapes.

Although natural geographical properties often vary continuously and randomly in space and over time, the pattern and scale of their variation is not readily apparent. A key issue in handling spatially distributed data is to determine the scale and the pattern of variation of the continuous spatial variables involved, and to estimate or interpolate values optimally at sites and locations that have not been visited and investigated. Geostatistics aims at providing quantitative descriptions of the distribution of natural variables in space or in both space and time. A number of user-friendly geostatistical tools have been developed recently. The variogram is a measure of how quickly, on the average, things change on a spatial scale. An experimental

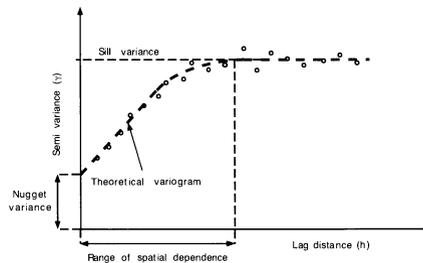


Figure 14.4: A theoretical variogram, the parameters of which - the sill variance, the range, and the nugget variance - describe the variation in the general characteristics involved.

variogram provides an estimate of a regional variogram or of a theoretical variogram, pertaining to land in the region sampled. A regional variogram is continuous, whereas an experimental variogram is discontinuous. To obtain an approximation to the theoretical variogram, one needs to fit a smooth curve or surface through the experimental points obtained and treat point-to-point fluctuations as sampling effects (Oliver 1999). The final use of a theoretical variogram is in setting up a kriging system in order to be able to perform a kriging interpolation. This can be done by creating an experimental variogram from the data set available and replacing it by a theoretical variogram. Variograms can usually be defined by three properties: sill, range and nugget (Fig. 14.4). The sill is where the variogram (γ) increases up to a maximum and where the values show a maximum dissimilarity with the remainder of the region. The range indicates at which lag distance (h) dissimilarity stops increasing: it marks the limit of spatial dependence. Places separated by a distance greater than this are considered to be spatially independent. At a lag of zero the semivariance should theoretically be zero, as well. Usually an adjusted smooth curve has a positive intercept at the ordinate. This feature is well known in mining and is called the 'nugget variance' or the nugget-effect. The nugget-effect arises due to the erratic behaviour of a regionalized variable on a very small scale, the variogram from zero to the nugget encompassing a distance shorter than the sampling distance. The nugget is sometimes referred to as white noise (Cressie 1991). One of the goals of the geochemical investigation at Asa was to estimate the values of the regionalized variables such as soil properties, as well as different modelled outputs. Geostatistical procedures for doing this are known generally as *kriging* (Oliver 1999). Kriging is a geostatistical estimation technique providing optimal unbiased spatial predictions or estimates on the basis of a linear combination of the values sampled.

It is a method involving a local weighted averaging of the values observed (eq. 14.8).

$$\widehat{Z} = \sum_{i=1}^N \lambda_i z(x_i), \tag{14.8}$$

where λ_i are the weights. If one wants an unbiased estimate, one sets the weights to 1:

$$\sum_{i=1}^N \lambda_i = 1 \tag{14.9}$$

Through Kriging we produced interpolated maps of the mineralogy, texture and moisture and of such modelled output values as the weathering rates and the sustainable yield of Ca, Mg and K. At the present stage, which is still premature, no geostatistical trend analyses or anisotropy analyses have been carried out on the data set and no consideration been taken of possible outliers. Geostatistical analysis of this sort will be undertaken later when more adequate data on the uncertainty of the various parameters are available. The variograms of the different parameters, the data input to the model, and the output values are very dissimilar. This is to be expected since the data are of different origin and are dissimilar in their uncertainty. The data itself also has a wide range of expected distribution patterns. The distribution of different physical and geological parameters of the soil can be expected to have a different distribution pattern than that for the deposition of cations and or for spatial variations in temperature. This is also something that is reflected very clearly in the different variograms. The variograms of the mineralogy, which are important input parameters Jönsson et al. (1995), show an exponential or spherical behaviour with a clear nugget effect (Fig. 14.5). The variograms modelled from the output values differ, most of the variograms of the output values behaving in accordance with a power law, but some showing exponential behaviour instead (Fig. 14.5). The differences in the behaviour of the different theoretical variogram models are difficult to explain. Also, the various input parameters differ in the "weights" assigned to them depending on the scenario one models. Thus, if an input parameter that has a strong influence on the output value modelled shows strong exponential behaviour, this may possibly reflect the behaviour of the variogram in terms of the output value in question.

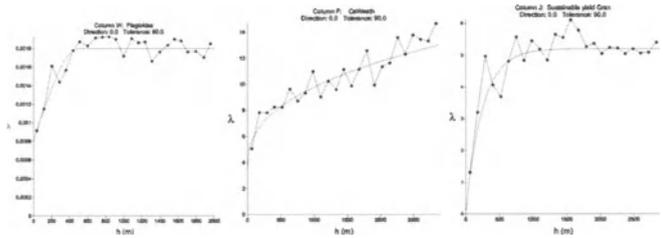


Figure 14.5: *Experimental variogram for the modelled plagioclase content of the soil. The modelled variogram shows spherical behaviour with a maximum lag distance of 2,000 m and a lag width of 80 m. The range is 500 m. The modelled variogram for Ca-weathering shows power of law behaviour with a maximum lag distance of 2,900 m and a lag width of 116 m. The modelled variogram for sustainable yield shows exponential behaviour with a maximum lag distance of 2,900 m and a lag width of 116 m. The effective range is 750 m.*

| Process | Value |
|------------------------------------|---|
| Productivity | 9.7 m ³ stemwood ha ⁻¹ yr ⁻¹ |
| Royal Forest Park area | 3,288 ha |
| Forest Research Park area | 1,200 ha |
| Norway Spruce | 60% |
| Scots Pine | 25% |
| Birch | 14.5% |
| Beech and Oak | 0.5% |
| Soil organic matter amount, 0-0.5m | 50-100 kg m ⁻² |
| New organic matter | 1.2-2 kg m ⁻² yr ⁻¹ |
| Soil nitrogen amount, 0-0.5m | 0.3-0.6 kg m ⁻² |
| Nitrogen flux in the soil at 0.5m | 0.015 kg m ⁻² yr ⁻¹ |
| Soil moisture content at 0.3m | 0.2 m ³ water m ⁻³ soil |
| Soil pH, O-layer | 4.3 |
| Soil pH, E-layer | 4.6 |
| Soil pH, B-layer | 5.2 |
| Soil C/N ratio | 25-30 kg/kg |
| Soil temperature | 6°C |
| Acidity deposition | 10-13 kg S ha ⁻¹ yr ⁻¹ |
| Nitrogen deposition | 14-16 kg N ha ⁻¹ yr ⁻¹ |
| Annual precipitation | 8000 m ³ water ha ⁻¹ yr ⁻¹ |
| Annual percolation | 6500 m ³ water ha ⁻¹ yr ⁻¹ |
| Annual runoff | 4500 m ³ water ha ⁻¹ yr ⁻¹ |
| Moose population | 0.15 moose equivalents ha ⁻¹ |
| Moose/deer ratio | 0.2 |

Table 14.2: *Asa case study. The total area surveyed included both the Asa Forest Research Park of 12 km² and the Asa Royal Forest Park of 36 km². A total of 366 soil sampling sites have been established.*

| | | |
|-------------|---|--|
| K-Feldspar | = | $\max(0, 7.46 * K_2O - 0.75 * Na_2O)$ |
| Plagioclase | = | $\max(0, 11.1 * Na_2O - 0.22 * K\text{-Feldspar})$ |
| Apatite | = | $2.24 * P_2O_5$ |
| Hornblende | = | $\max(0, 6.67 * CaO - 3.67 * Apatite - 0.2 * Plagioclase)$ |
| Biotite | = | $\max(0, 3.85 * MgO - 0.39 * Hornblende)$ |
| Epidote | = | $\max(0, 0.1 * Hornblende + 0.03 * Plagioclase - 0.3)$ |
| Muscovite | = | $\max(0, 0.5 * K_2O - 0.05 * Na_2O)$ |
| Quartz | = | $SiO_2 - 0.63 * Plagioclase - 0.68 * K\text{-Feldspar} - 0.38 * Muscovite - 0.33 * Chlorite - 0.45 * Hornblende - 0.42 * Epidote$ |
| Al-residual | = | $Al_2O_3 - 0.1 * Plagioclase - 0.1 * K\text{-Feldspar} - 0.26 * Muscovite - 0.09 * Chlorite - 0.01 * Hornblende - 0.025 * Epidote$ |
| Delta | = | Quartz + Plagioclase + K-Feldspar + Muscovite + Chlorite + Hornblende + Epidote + Apatite |
| Vermiculite | = | $\min(Al\text{-residual}, \max(0, 100\% - Delta))$ |

Table 14.3: *The UPPSALA normalization routine used at Asa Forest Research Park for calculation of soil mineralogy. The mineralogy was also adjusted for observed loss on ignition (LOI). Input is elemental contents of oxides in % of weight.*

| Substance | Wet deposition | Dry deposition |
|-------------------|----------------|----------------|
| Sulphur | 33.3 | 33.3 |
| Nitrate | 26.0 | 13.0 |
| Ammonium | 24.4 | 12.0 |
| Chloride | 29.6 | 21.0 |
| Natrium | 25.1 | 13.0 |
| Calcium | 12.4 | 20.0 |
| Magnesium | 7.6 | 10.0 |
| Kalium | 3.6 | 10.0 |
| Effective acidity | 64.6 | 38.3 |
| Titration acidity | 15.8 | 14.3 |

Table 14.4: *Parameters used for distributing the deposition over the area in $mEq\ m^{-2}\ yr^{-1}$.*

Deposition and climate

The deposition was calculated as follows from the average values for the entire park:

$$D = (D_{wet} + D_{dry} \cdot \frac{Y}{Y_M}) \cdot \frac{P}{P_M} \tag{14.10}$$

where Y is the forest yield, Y_M is the average yield ($9.9 \text{ m}^3 \text{sk ha}^{-1} \text{ yr}^{-1}$), P is rainfall and P_M is the average rainfall for the entire area (precipitation is $0.816 \text{ m}^3 \text{ m}^{-2}$, the amount reaching the ground surface is $0.618 \text{ m}^3 \text{ m}^{-2}$, average runoff Q, is $0.468 \text{ m}^3 \text{ m}^{-2}$). The distribution of rainfall over the area was obtained by:

$$P = P_0 + 1.04 \cdot (z - 226) \tag{14.11}$$

where $P_0=262 \text{ m}^3 \text{ m}^{-2}$ and z is elevation above sea level in meters. The loss of precipitation through evapotranspiration is scaled according to the temperature dependence of the forest growth:

$$EV = EV_0 \cdot 10^{\frac{3500}{280.15} - \frac{3500}{T}} \tag{14.12}$$

where $EV_0 = 0.150$ and the reference temperature is 7.53°C . The runoff was calculated as the precipitation surplus:

$$Q = P - EV \tag{14.13}$$

The precipitation is shown in Figs. 14.10. At Asa the annual average soil temperature is 7.53°C and the average annual air temperature is 6.5°C . The temperature on the ground, considering elevation and degree of exposure to the sun, was estimated using:

$$T = T_0 - 0.05 \cdot (z - 226) + 0.5 \cdot \sin(S) \tag{14.14}$$

where T is the temperature, T_0 is the reference temperature of 7.43 degrees Centigrade, and z is the elevation above sea level. S is the corrected direction s:

$$S = s + 120^\circ. \tag{14.15}$$

Nutrient uptake

The nutrient contents were taken from a synthesis of biomass contents. The uptake to the biomass was calculated from the forest yield as follows:

$$BC_{U,j} = Y \cdot \rho_j \cdot 10 \cdot \left(\frac{x_{Ca,j}}{E_{Ca}} + \frac{x_{Mg,j}}{E_{Mg}} + \frac{x_{K,j}}{E_K} \right) \tag{14.16}$$

$$N_{U,j} = Y \cdot \rho_j \cdot 10 \cdot \left(\frac{x_{N,j}}{E_N} \right) \tag{14.17}$$

where j is the biomass fraction (j=stemwood, branches, bark, leaf, root), x_i is weight fraction in % of dry weight, and E_i is the equivalent weight of the ions; Ca=20, Mg=12.15, K=40, N=14. The ecosystem nutrient uptake efficiency was defined as:

$$E_R = 100 \cdot \left(1 - \frac{L_{min}}{W + D - L_{phys}} \right) \tag{14.18}$$

where L_{min} is the lower range of the leaching that occur from the bottom of the root zone in the forest system because of root apparatus limitations, and L_{min} is the leaching caused by physiological inability to take up base cations below a certain concentration. The amount in the divisor represents all base cations potentially available to the plant, the leaching term above, what the system actually could not capture. Efficiencies were estimate for Asa on the basis of the leaching data available. This efficiency has two components; partly the physiological limitation given by the lowest solute concentrations at which uptake will take place and secondly the uptake efficiency determined by the degree of soil penetration by the roots. These two factors are multiplicative and approximately equally important at Asa.

| Forest vegetation type | Rooting depth | Root system estimated efficiency |
|------------------------|---------------|----------------------------------|
| Norway Spruce | 0.35 | 77 |
| Scots pine | 0.50 | 88 |
| Silver Birch | 0.60 | 92 |
| Oak | 0.80 | 97 |
| Beech | 0.80 | 97 |
| Maple | 0.75 | 95 |
| Ash | 0.75 | 95 |
| <hr/> | | |
| Spruce-Beech | 0.65 | 95 |
| Pine-Oak | 0.75 | 95 |
| Spruce-Birch | 0.45 | 85 |
| Mixed broadleaves | 0.75 | 95 |

Table 14.5: *Estimated average rooting depth in different types of forest stands and approximate root system base cation capture efficiency.*

We have back-calculated root efficiencies for Norway spruce, European Beech and mixed Beech-Spruce forest, using the data taken from Asa Forest Research Park. The Norway spruce forest ecosystem has a 77-80% base cation uptake efficiency at a rooting depth of 0.3 m, for a mixed forest the uptake efficiency is 80-85% and for stands dominated by beech or oak the uptake efficiency is approximately 92-97%. The observations suggest that the lower limit set by physiological limitations and efficiency limitations of the root apparatus is 85% on the average, the standard deviation from this is only $\pm 6\%$. The root system base cation capturing efficiency was estimated by an equation based on the observation that a Beech stand has approximately 92-97% base cation capture efficiency and a Norway spruce stand approximately 75-80% base cation capture efficiency. For Norway spruce the average rooting depth was estimated to 0.3-0.4 m, for an average European beech or Oak forest it was

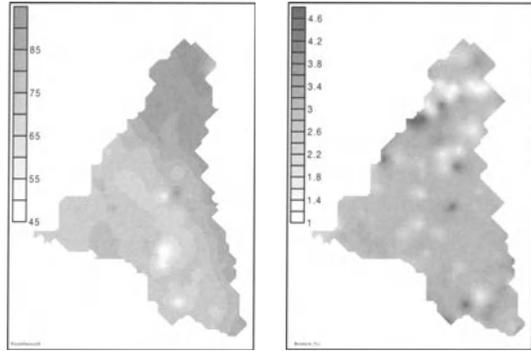


Figure 14.7: *At the left is a map of the calculated root base cation capture efficiency in the Asa Forest Research Park. Root efficiency and of variable rooting depth are basic to the hypotheses investigated in the SUFOR programme and to an understanding of the differences in sustainability performance of different tree species under field conditions. At the right the observed soil moisture at Asa is shown. The soil moisture affect the root uptake efficiency. Average root efficiency at Asa has a normal distribution of 75-95%, with the average around 85%.*

| | | | | | |
|------------------|------|------|------|------|--------|
| Norway Spruce | Ca | Mg | K | N | ρ |
| Stemwood | 0.14 | 0.02 | 0.07 | 0.10 | 450 |
| Branch | 0.23 | 0.08 | 0.63 | 0.50 | 450 |
| Needle | 0.38 | 0.06 | 0.68 | 1.00 | 300 |
| Scots Pine | Ca | Mg | K | N | ρ |
| Stemwood | 0.10 | 0.02 | 0.05 | 0.08 | 500 |
| Branch | 0.20 | 0.05 | 0.21 | 0.30 | 450 |
| Needle | 0.38 | 0.08 | 0.51 | 0.90 | 300 |
| European Beech | Ca | Mg | K | N | ρ |
| Stemwood | 0.11 | 0.03 | 0.10 | 0.10 | 700 |
| Branch | 0.24 | 0.03 | 0.13 | 0.25 | 600 |
| Leaf | 0.41 | 0.08 | 0.90 | 2.00 | 300 |
| Silver Birch | Ca | Mg | K | N | ρ |
| Stemwood | 0.16 | 0.04 | 0.08 | 0.18 | 600 |
| Branch | 0.41 | 0.06 | 0.18 | 0.63 | 600 |
| Leaf | 0.90 | 0.28 | 0.96 | 1.80 | 300 |
| Bark | 0.39 | 0.03 | 0.14 | 0.27 | 300 |
| Clear-cut debris | 0.04 | 0.01 | 0.04 | 0.33 | 300 |
| European Oak | Ca | Mg | K | N | ρ |
| Stemwood | 0.11 | 0.02 | 0.13 | 0.18 | 750 |
| Branch | 0.40 | 0.03 | 0.08 | 0.09 | 650 |
| Leaf | 1.00 | 0.20 | 0.30 | 1.80 | 300 |

Table 14.6: Parameters used for assessing nutrient content in different parts of different tree species. All numbers are % of dry weight. ρ has units of kg m^{-3} .

set at 0.8 m. The following equation was adopted:

$$E_{\text{Root}} = 0.23z^{0.35} \quad (14.19)$$

For all the trees, the physiological uptake limitation allow the root to take up approximately 95-97% of all the base cations that come into physical contact with the root when no disturbances from acid soils are present. Acidification of soils can change the physiological efficiencies drastically to the degree where bulk uptake the tree can become severely restricted.

| Ca | Mg | K | Na | Soil depth | Comment | |
|------|------|------|------|------------|--|-----------------------|
| 1.57 | 0.52 | 0.08 | 1.55 | 50cm | Intensive plantation 15-20 m ³ sk ha ⁻¹ yr ⁻¹ | |
| 0.87 | 0.63 | 0.06 | 1.52 | 50cm | | |
| 1.22 | 0.75 | 0.02 | 1.64 | 50cm | | |
| 0.41 | 0.35 | 0.08 | 1.01 | 50cm | | |
| 1.82 | 0.56 | 0.08 | 2.05 | 50cm | | |
| 0.81 | 0.47 | 0.08 | 1.35 | 50cm | | |
| 1.37 | 0.81 | 0.08 | 2.17 | 50cm | | |
| 0.41 | 0.31 | 0.10 | 1.06 | 50cm | | |
| 0.72 | 0.72 | 0.21 | 5.48 | 25cm | | Extensive Forestry |
| 1.06 | 1.02 | 0.23 | 9.6 | 50cm | | |
| 1.22 | 0.59 | 0.38 | 3.53 | 25cm | 7-10 m ³ sk ha ⁻¹ yr ⁻¹ | |
| 1.58 | 0.85 | 0.17 | 6.53 | 50cm | | |
| 0.82 | 0.63 | 0.14 | 5.88 | 25cm | | |
| 1.05 | 0.79 | 0.13 | 7.8 | 50cm | | |
| 0.65 | 0.83 | 0.12 | 8.9 | 25cm | | |
| 0.79 | 0.81 | 0.20 | 8.3 | 50cm | | |
| 2.08 | 1.19 | 0.35 | 6.54 | 50cm | Clear-cut 1 m ³ sk ha ⁻¹ yr ⁻¹ | |
| 2.68 | 1.23 | 0.24 | 8.53 | 50cm | | |

Table 14.7: Observed rooting zone leaching in the Asa Research Park, expressed in mg l⁻¹.

| Ca | Mg | K | Na | Comment |
|------|-------|--------|------|--|
| 1.06 | 0.55 | 0.073 | 1.03 | Intensive, 15-20 m ³ sk ha ⁻¹ yr ⁻¹ |
| 1.00 | 0.78 | 0.20 | 6.98 | Extensive, 7-10 m ³ sk ha ⁻¹ yr ⁻¹ |
| 2.38 | 1.21 | 0.30 | 7.50 | Open land, 1 m ³ sk ha ⁻¹ yr ⁻¹ |
| 1.03 | 0.63 | 0.12 | 3.35 | mg l ⁻¹ , Average leaching values in Asa |
| 0.4 | 0.3 | 0.06 | - | mg l ⁻¹ , Lower limit for leaching |
| 0.02 | 0.025 | 0.0015 | - | mEq l ⁻¹ , Lower limit for leaching |
| 0.2 | 0.1 | 0.04 | - | mg l ⁻¹ , Physiological limit for uptake |
| 0.01 | 0.01 | 0.001 | - | mEq l ⁻¹ , Physiological limit for uptake |
| 1.0 | 0.5 | 0.08 | - | Used in budget calculations, mg l ⁻¹ |
| 0.05 | 0.041 | 0.002 | - | Used in budget calculations, mEq l ⁻¹ |

Table 14.8: Summary of leaching in different forest types of differing growth intensity. Concentrations are in mg l⁻¹

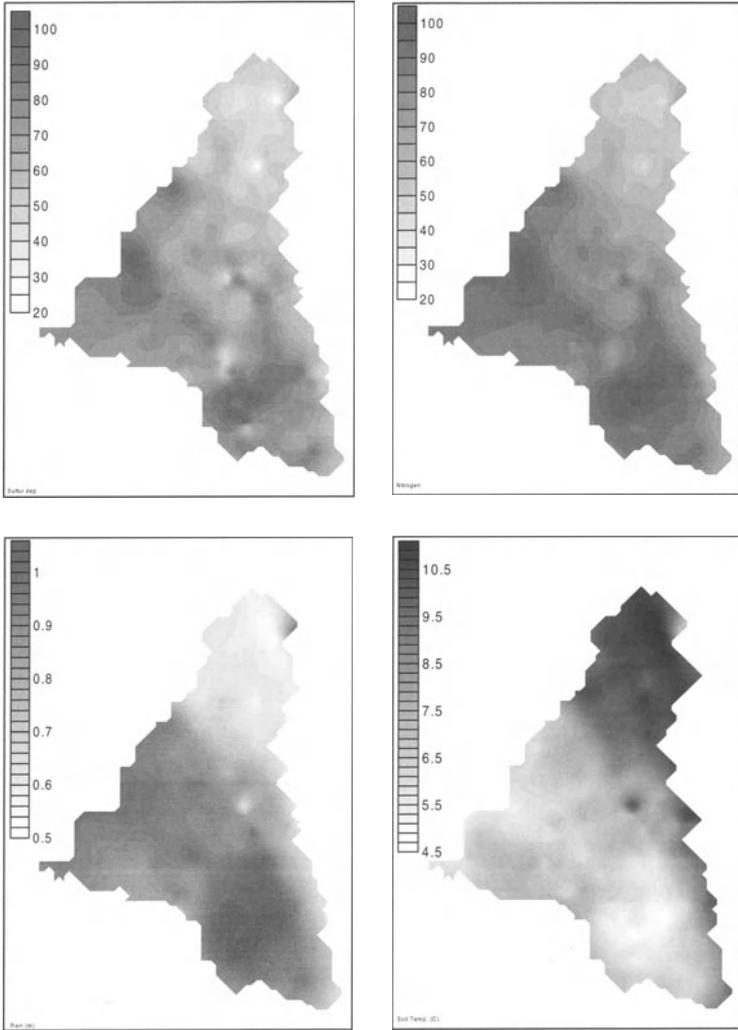


Figure 14.6: *Estimated sulphur and nitrogen deposition, precipitation and soil moisture to a full grown needle forest stand at Asa.*

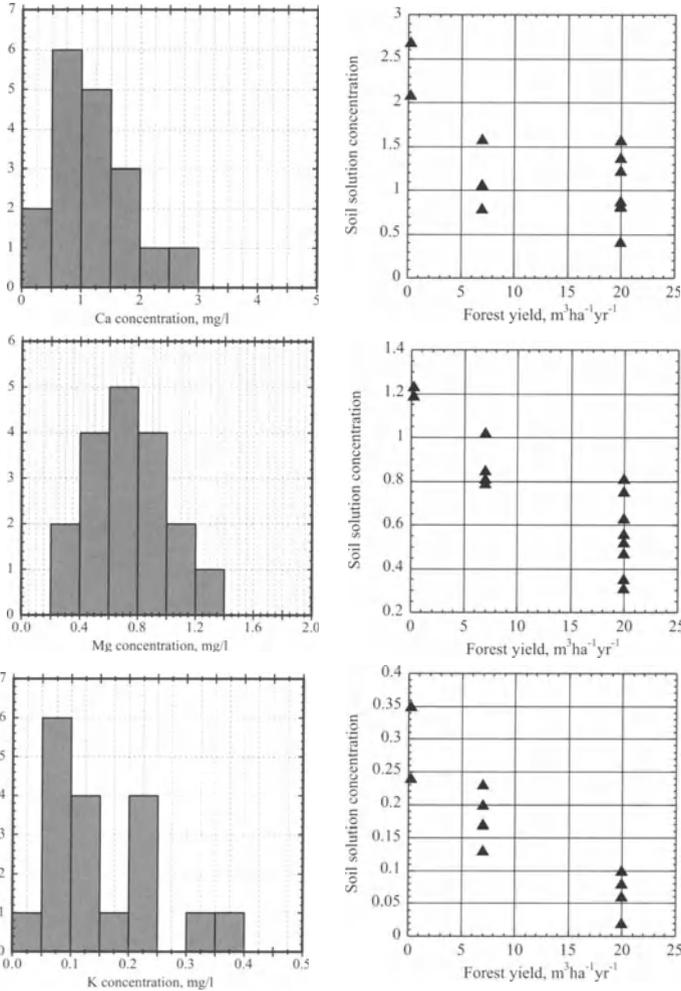


Figure 14.8: To the left; Frequency distribution of observed concentrations of Ca and Mg in the leaching water at the 50 cm level in the soil. To the right; Dependence of concentration in the leachate at the 0.5 meter depth in forests soils as related to growth intensity in $m^3 \text{ sk ha}^{-1} \text{ yr}^{-1}$. Upper left; Calcium, upper right; Magnesium and lower left; Potassium. With growing intensity, the forest will take up more and more of the available base cations. The data suggest uptake efficiencies from approximately 77% for Norway spruce with 0.3-0.35 m rooting depth to approximately 95% in deep rooted deciduous forest (0.65-0.8 meter).

| Soil layer | Soil depth | Spruce | Pine | Birch | Beech | Oak |
|------------|------------|--------|------|-------|-------|-----|
| O/A | 0-0.10 | 55 | 55 | 40 | 20 | 15 |
| E | 0.10-0.20 | 26 | 25 | 25 | 25 | 25 |
| B | 0.20-0.35 | 13 | 13 | 15 | 30 | 25 |
| B/C | 0.35-0.55 | 6 | 4 | 10 | 15 | 20 |
| C | 0.55-0.80 | - | 2 | 5 | 10 | 10 |
| | 0.80-1.00 | - | - | - | - | 5 |

Table 14.9: Approximate root distributions as weight % for the different tree species to be considered in the SUFOR study. The data was taken from German studies from Solling and in Swedish studies.

| Soil layer | Soil depth | Spruce Ca, Mg | Spruce K, N | Deciduous Ca, Mg | Deciduous K, N |
|------------|------------|---------------|-------------|------------------|----------------|
| O/A | 0-0.10 | 40 | 60 | 20 | 50 |
| E | 0.10-0.20 | 30 | 30 | 25 | 25 |
| B | 0.20-0.35 | 30 | 10 | 25 | 15 |
| B/C | 0.35-0.55 | - | - | 10 | 10 |
| C | 0.55-0.80 | - | - | 10 | - |
| | 0.80-1.00 | - | - | 10 | - |

Table 14.10: Approximate uptake distribution of Norway spruce and of deciduous stands. K and N uptake are assumed to follow the root distribution, the uptake being greater near the surface, Ca and Mg uptake are more strongly affected by the weathering rate, which has been shown to consistently increase with depth.

| Forest type | Ca | Mg | K |
|---------------------------------|-------------------------------------|-------------------------------------|------------------------------------|
| Weathering, Norway Spruce | 5.0 ^{1.5} _{9.8} | 0.9 ^{0.05} _{3.9} | 2.8 ^{0.9} _{6.0} |
| Weathering, Spruce-Beech; 50:50 | 15.3 ^{3.3} _{34.3} | 4.8 ^{0.28} _{24.8} | 7.7 ^{2.3} _{16.7} |
| Weathering, Beech | 20.1 ^{5.9} _{44.1} | 6.7 ^{0.4} _{32.9} | 9.7 ^{2.9} _{21.7} |
| Deposition | 32 ^{8.2} ₅₀ | 17.5 ⁵ ₂₇ | 13.5 ^{2.4} ₂₂ |
| Leaching | 18 ^{3.8} ₃₂ | 15 ^{2.7} ₂₆ | 1.0 ^{0.2} _{1.6} |

Table 14.11: Summary of the weathering rate calculations. The large number is the average, the lower case the maximum and the upper case the minimum. Below, comparable numbers for atmospheric deposition and leaching is shown, units are mEq m⁻¹yr⁻¹.

14.1.7 Biogeochemical mass balances at Asa

The sustainability assessments at Asa have focused on aspects of key parameters of the mass balance calculation. The method applied is to make a budget calculation for each cation separately. Taking deposition and weathering as the sources of base cations, and uptake and leaching as the sinks, the mass balance for each base cation becomes:

$$\Delta_i = W_i + D_i - U_i - L_i \quad (14.20)$$

where U is uptake and depend on the amounts removed by harvest from the system, D is deposition which partly depend on human activities, W is weathering and L is leaching which depend on water flux and acid deposition. The units are $\text{keq ha}^{-1}\text{yr}^{-1}$ in aall calculations and maps. i is any of the nutrients Ca, Mg, K and N. The calculation is made for each nutrient separately, and the sustainable yield is calculated according to Liebig's Law. In principle, this limitation applies to any nutrient and to water. Such a mass balance can be made for any bulk or trace nutrient. Sustainability is kept when the long-term average value of Δ is zero. For Swedish forestry, several studies indicate that the present harvesting volume of 70 million $\text{m}^3 \text{ year}^{-1}$ can be sustained by present weathering and cation deposition. Such studies tend to indicate that whole tree harvest is not sustainable, because the removal of nutrients then by far exceeds the supply. Work is continuing on this problem to investigate the details, but it illustrates that some of the things we do are sustainable, whereas other activities are clearly not. For nitrogen, we have a similar mass balance:

$$\Delta_N = \text{Deposition} + \text{Fixation} - \text{Immobilization} - \text{Uptake} - \text{Denitrification} \quad (14.21)$$

For many years, Swedish forest ecosystems have received more nitrogen than needed for sustainable growth, but with the development of the long range air pollutant protocols (UN-ECE/LRTAP convention), this may change. In areas far from densely populated regions in Europe, such as Sweden, the nitrogen deposition may decrease below what would possibly be optimal for the forest, as a result of current and future emission reduction plans. In the equation above, fixation and denitrification will be small for most practical purposes. There is a functional dependence between uptake and immobilization, generally immobilization will amount to 50% to 100% of the uptake. Thus, the uptake that can be supplied from deposition can be approximated by

$$N_{imm} = 0.2 \cdot N_{Litterfall} \quad (14.22)$$

The problem of acid rain has caused much attention to be focused on forest growth and forest management, and how these factors interact with different types of pollution stresses, even those not necessarily connected to acid rain. It has also been brought to general attention the necessity for the forest system to be sustainable as a production apparatus for biomass. Sustainable nutrient management was simply not necessary as long as production was low as compared to the maximal production capacity. In 1995, the Swedish forests produced an annual harvest of approximately 70 million $\text{m}^3 \text{ year}^{-1}$ timber from a net growth of almost 100 million $\text{m}^3 \text{ year}^{-1}$, this is more than ever before. The forest production is essential to the Swedish industrial production both as a source of raw material and revenue income. According to the 1990 Swedish Parliamentary Forest Commission, it should be promoted that the harvest increases to substantially more than 100 million $\text{m}^3 \text{ year}^{-1}$ by the year 2010. Industrial and cultural activities have in the time from the start of the industrialization in 1840 to present, resulted in steadily increasing emissions of nitric oxides and ammonia, increasing the atmospheric deposition of nitric acid and ammonium over the same time period. The increase in nitrogen available for growth occurred at the same time as forestry increased productivity in the forest two- to three-fold. The Swedish Forest Inventory can proudly point to the statistics showing that total forest growth has steadily increased since the beginning of their estimates in 1920. As the increase in growth appear stable and uninterrupted from 1920 to present, it may appear as if there is absolutely nothing to worry about.

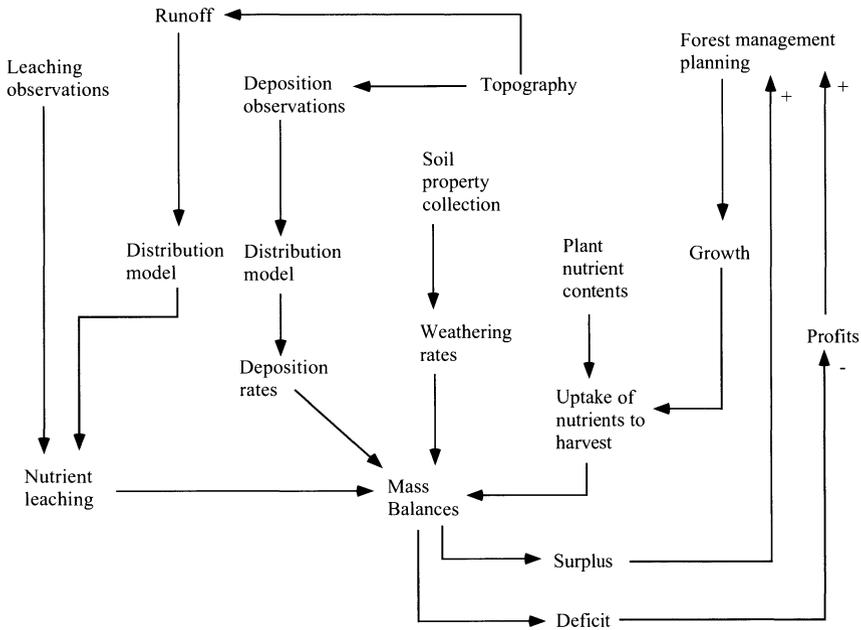


Figure 14.9: Flow diagram for assessment of base cation nutrient sustainability in forest management. The Assessment procedure required close cooperation between the different programme parts.

Historically, the forest management has been focused on increasing the productivity of the forest, with great success. This has resulted in more trees per unit land area, more efficient management of stands and promotion of tree age distributions that suit the planned harvesting needs. When very large volumes of biomass are removed from the forest system every year it is not longer self evident that there always will be sufficient supply of the necessary nutrients. In the soil, acidity is partly neutralized by alkalinity originating from weathering of soil minerals. In Swedish soils the weathering rate is low, and the present deposition of acidity mostly exceed the weathering rate. The residual acidity in the percolate (H^+ - and Al^{3+} -ions) will exchange with Ca, Mg and K adsorbed to organic matter. Through this process, the concentration of Ca, Mg and K in the soil solution will increase and the base saturation decreases. From 1840 to 1980 the acid deposition increased steadily, since 1980 it has decreased by approximately 40%. From 1900 to 1980 growth increased by better and more efficient management. During the same period the deposition acid also made an increasing amount of Ca, Mg and K available in the soil solution, "stealing" them from the base saturation. It goes almost without saying that the reservoir of exchangeable base cations should become very low after some time. When the base saturation reaches very low levels, it becomes very difficult to remove additional base cations, and the concentration of base cations in the soil solution will decrease.

From the perspective of sustaining high growth rates, problems may arise. The amount nitrogen available for tree uptake remains high, but the amount of Ca, Mg and K decrease. The plant requires relatively fixed ratios of available nitrogen:base cations in order to make needles, stem and roots, and the elasticity is not very large. Trees are have evolved to be adapted to a situation where nitrogen is always very limited, and where the best competitiveness is gained by taking it up as efficiently as possible. When the trees cannot match all the nitrogen taken

| Parameter | Unit | Source |
|--|-----------------------------------|---------------------------|
| Soil bulk density | kg m ⁻³ | From default |
| CO ₂ pressure | times ambient | From default |
| Dissolved organic carbon | mg l ⁻¹ | |
| Soil profile water uptake distribution | % of total | Default root distribution |
| Soil profile Mg+Ca+K uptake distribution | % of total | Default root distribution |
| Soil profile N uptake distribution | % of total | Default root distribution |
| Al equilibrium constant | kmol ² m ⁻³ | Default values |
| Soil depth mineralogy variations | | From data |
| Thickness of layers | | From data |

Table 14.12: *Input data taken from default values.*

up with the necessary amount of Mg, Ca or K to make tree parts, nutrient regulatory problems may arise because of the conflicting signals. Less needles can be sustained the following year when the available supply of base cations fall below the current need set by present growth rate and needle mass, eventually growth have to adapt to less needles. In the long term there will be no way to avoid Liebig's law (growth is limited by the nutrient in least supply). In due time any site must follow the principles of mass conservation. For any forest, a mass balance for any nutrient can be made. The following is typical of how such a calculation can be made; typical numbers for magnesium were taken from the Swedish forest Inventory made at Asa Research Park, South Sweden. The deficit must be taken from some type of internal reservoir, generally from the base saturation. This occurs by ion exchange, but may also occur by temporal net mineralisation of the organic matter. The annual deficit is 2% of the adsorbed magnesium. This is too small to detect in soil analysis, but the mass balance can easily detect the imbalance. The resistance to acidification has been estimated for Swedish soils, this is called critical loads. The critical loads are used internationally to negotiate pollution emission reductions. It has been discovered that the critical load is dependent on the forest management and harvest intensity. Whereas some methods of intensive forest exploitation lower the resistance to acid pollution, other methods may possibly increase the resistance and thus improve the situation.

The present site yield index (Swedish; Bonitet) does not coincide with the amount of base cations available from weathering. The present site yield index is influenced by nitrogen availability, which for a large part of the area is not long term limiting. An alternative site yield index need to be developed, based on base cations, and the minimum of the two must be searched for. Whole tree harvest is already practiced in a large part of Southern Sweden. This is presently done without return of base cation nutrients. This implies that many these landowners will be quickly consuming the nutrient resources of the soil and probably significantly reducing their fertility. This will also have repercussions on future forest property value. An important question to answer is what happens when the trees have more than needed of nitrogen but less than needed of Mg or K? If the amount Mg that can be taken up is less than the need of the present needle mass, then the excess needles will be cast off. The effect may be short term offset if the tree can mobilize internal sources of Mg, but such stores are relatively small compared to the need during the entire rotation period. Needle loss would according to this be a symptom of nutrient stress. If K is lacking, discoloration follows, but later also needle loss. The observed deficits in Asa amounts vary from 0 to 0.25 keq ha⁻¹yr⁻¹, the amount 0.25 keq ha⁻¹yr⁻¹ is corresponding to 50-60 SEK ha⁻¹yr⁻¹ in limestone or 60-80 SEK ha⁻¹yr⁻¹ in full fertilizer. This would amount to 3,500-10,500 SEK ha⁻¹ over a rotation period. The projected profit is in the range of 250-1,500 SEK ha⁻¹yr⁻¹ for a Norway spruce stand, and the nutrient addition cost would be 5-20% of the profit.

| Parameter | Unit | Source |
|---------------------------------------|-----------------------------------|--------------------------|
| Site identification | | |
| Stand number | | |
| Area of stand | | ha |
| Longitude | | |
| Latitude | | |
| Temperature | | °C |
| Precipitation | m ³ /m ² yr | From data and model |
| Runoff from rootzone | m ³ /m ² yr | From data and model |
| Sulphur deposition | keq/m ² yr | |
| Nitrate deposition | keq/m ² yr | |
| Chloride deposition | keq/m ² yr | |
| Ammonium deposition | keq/m ² yr | |
| Sodium deposition | keq/m ² yr | |
| Calcium deposition | keq/m ² yr | |
| Kalium deposition | keq/m ² yr | |
| Sulphur deposition | keq/m ² yr | |
| Vegetation type | | Species |
| Mg+Ca+K net uptake | % of total max | From yield and model |
| N net uptake | % of total max | From yield and model |
| Mg+Ca+K total uptake | % of total max | From yield and model |
| N total uptake | % of total max | From yield and model |
| K-feldspar | % of total | From data |
| Plagioclase | % of total | From data |
| Hornblende | % of total | From data |
| Pyroxene | % of total | From data |
| Epidote | % of total | From data |
| Calcite | % of total | From data |
| Biotite | % of total | From data |
| Muscovite | % of total | From data |
| Chlorite | % of total | From data |
| Vermiculite | % of total | From data |
| Apatite | % of total | From data |
| Anorthite to Albite ratio in feldspar | | |
| Specific surface area | Classes | From classification data |
| Soil type | class | From data |
| Moisture class | 1-6 | From classification data |
| 1. layer thickness | m | From data |
| 2. layer thickness | m | From data |
| Rooting depth | m | From vegetation type |

Table 14.13: *Input data specification for the regionalized version of PROFILE.*

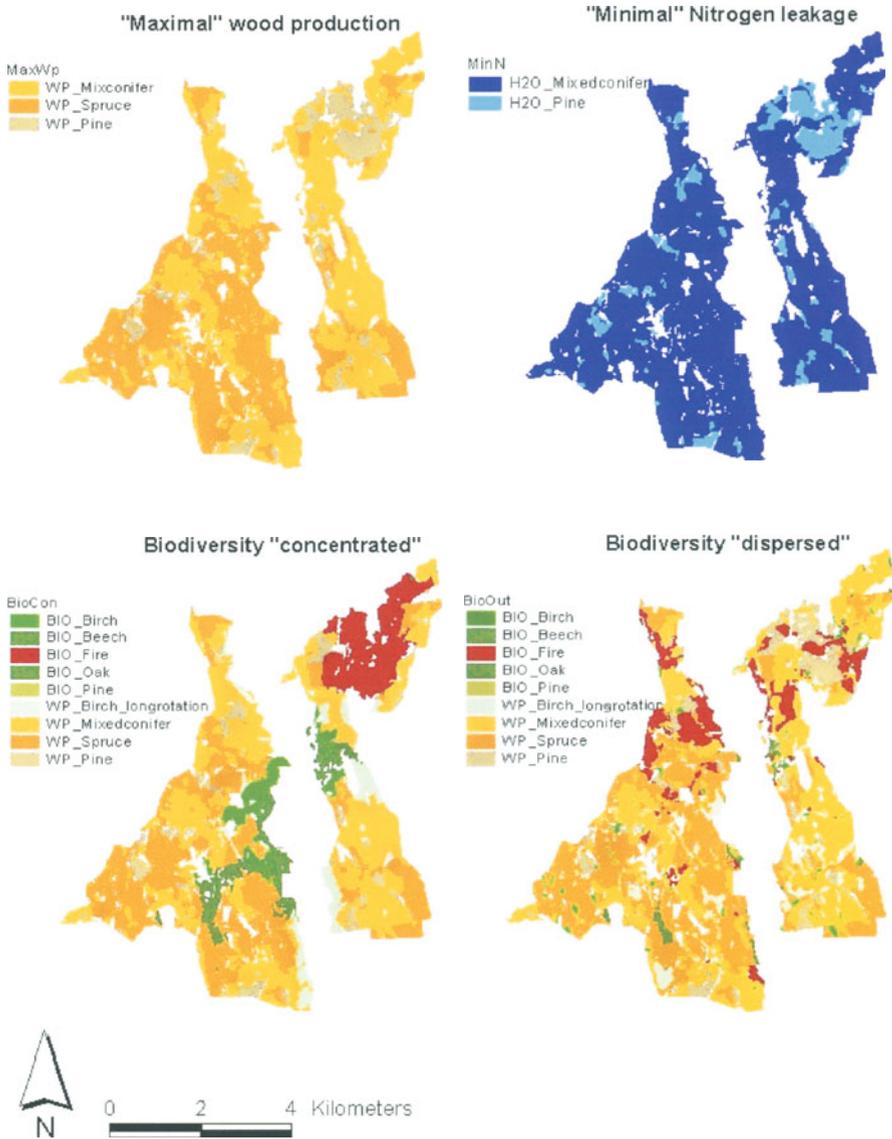


Figure 14.10: *Different production scenarios tried out at Asa in Chapter 13 on assessing sustainability aspects.*

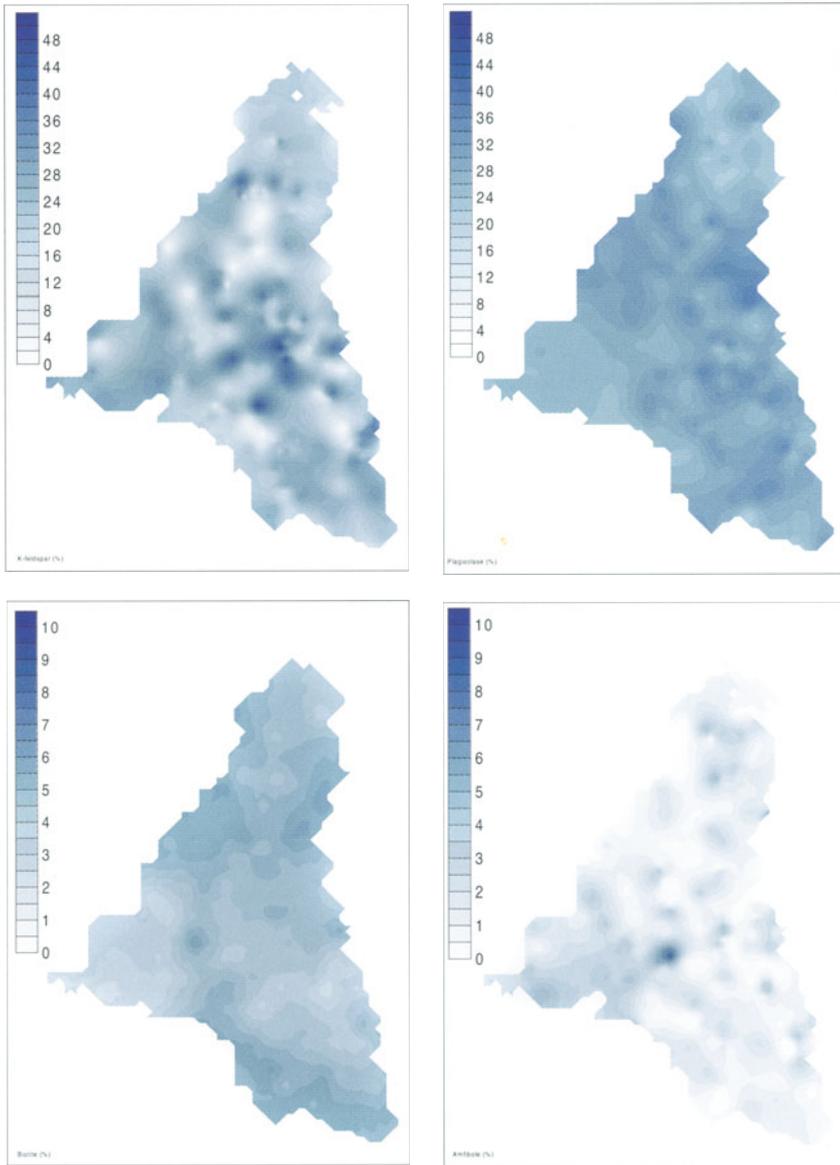


Figure 14.11: *Estimated soil mineralogy at Asa, in % weight, used in Chapter 14. Shown are K-feldspar (top, left), plagioclase (top, right), hornblende (bottom, left) and epidote (bottom, right). In total, 12 minerals were calculated for the Asa soils.*

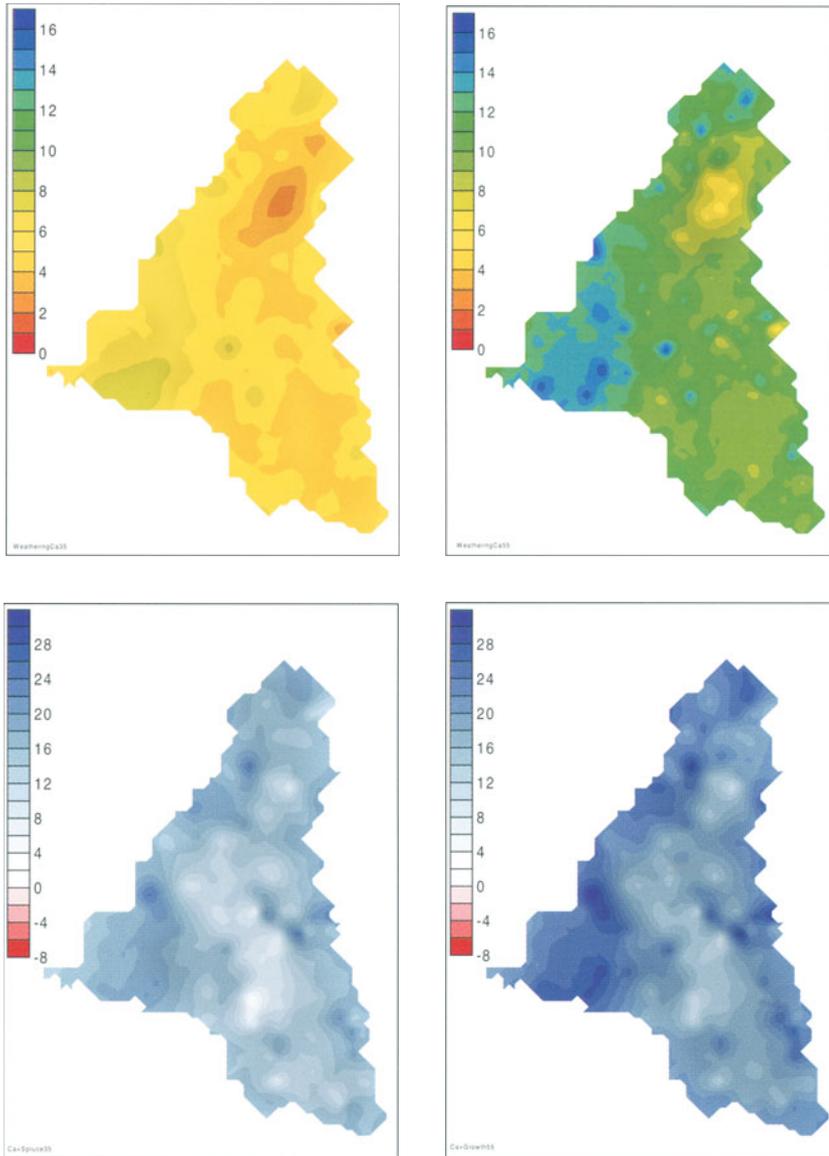


Figure 14.12: Calcium weathering for Norway Spruce monoculture assuming 0.35 m rooting depth (top, left) and in a mixed stand (top, right). Mass balance residual for Ca for Norway spruce monoculture (bottom, left) and for mixed forest (bottom, right). Units are mEq m⁻² yr⁻¹.

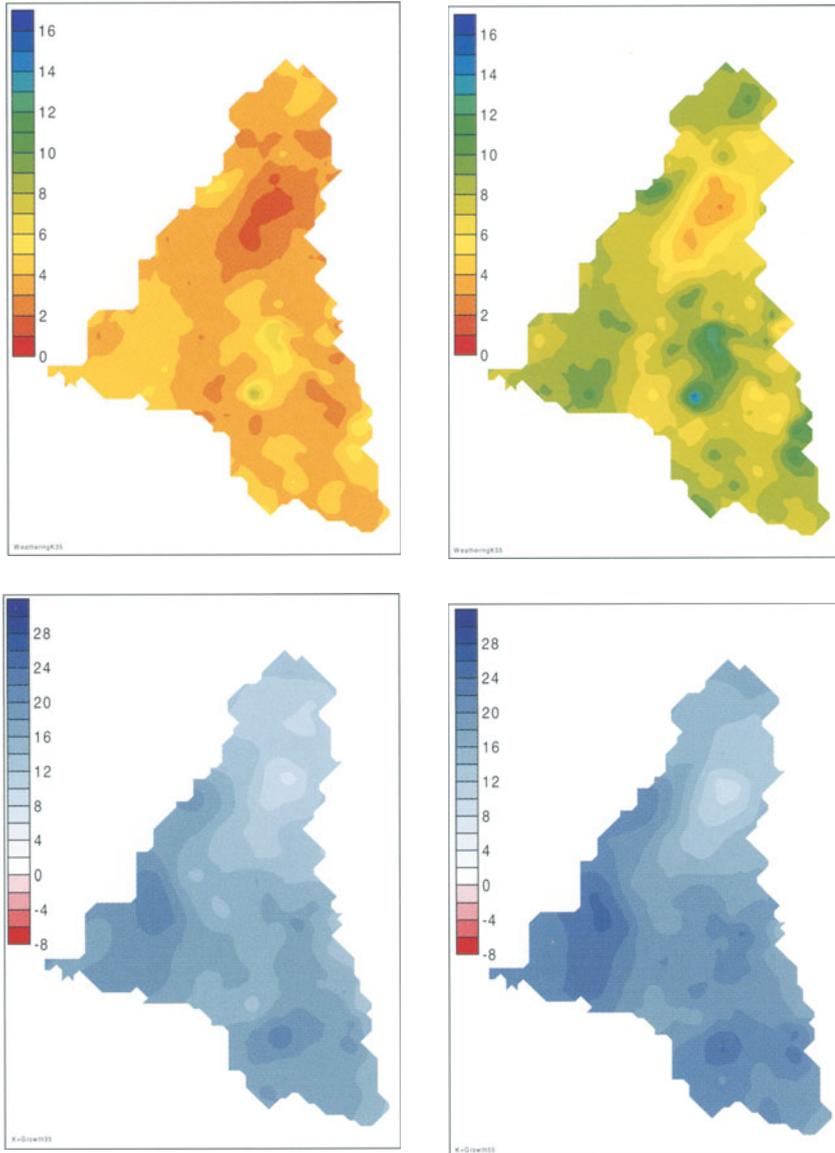


Figure 14.13: Potassium weathering for Norway Spruce in monoculture assuming 0.35 m rooting depth (top, left) and in a mixed stand (top, right). Mass balance residual for Potassium in a Norway spruce forest (bottom, left) and for mixed forest (bottom, right). The red areas show a deficit. Units are $\text{mEq m}^{-2} \text{yr}^{-1}$.

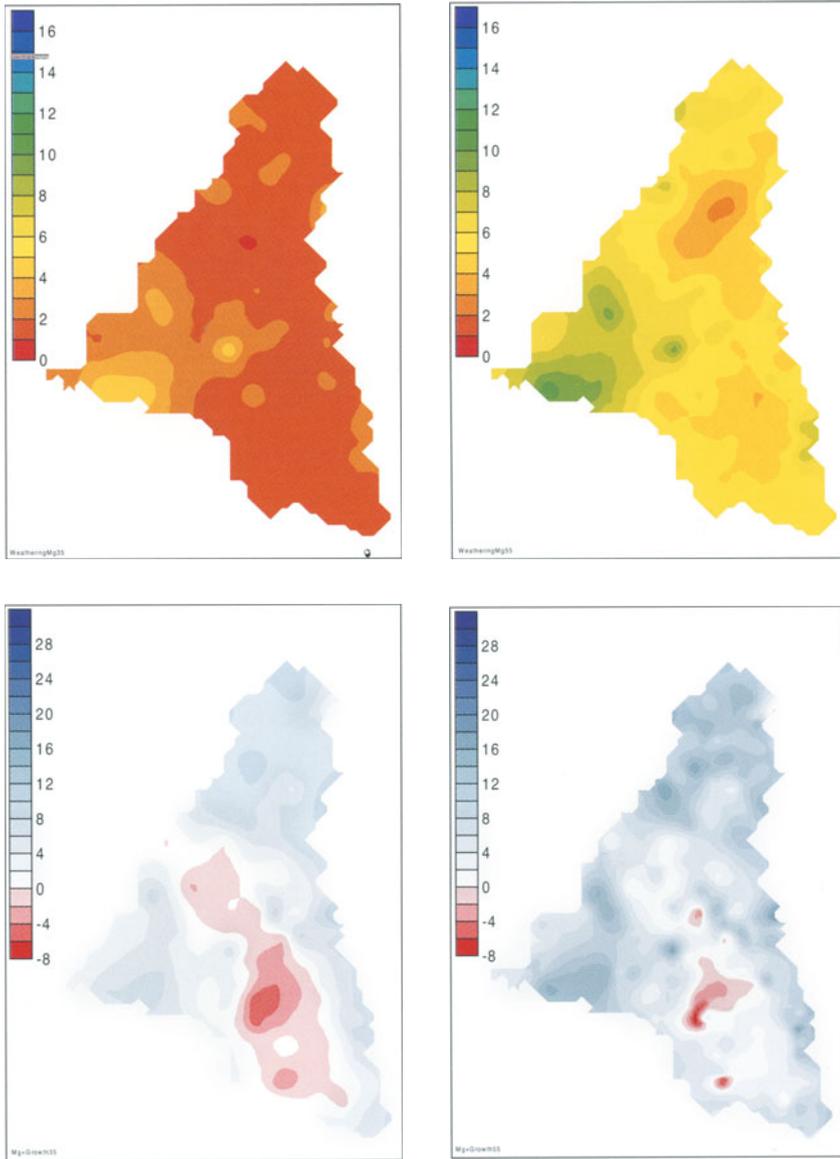


Figure 14.14: *Magnesium weathering for Norway Spruce in monoculture assuming 0.35 m (left, top) and in a mixed stand (right, top). Mass balance residual for Magnesium in Norway spruce forest (left, bottom) and for mixed forest (right, bottom). The red areas show a deficit and would need additions of magnesium to be sustainable for harvest. Units are $mEq\ m^{-2}\ yr^{-1}$.*

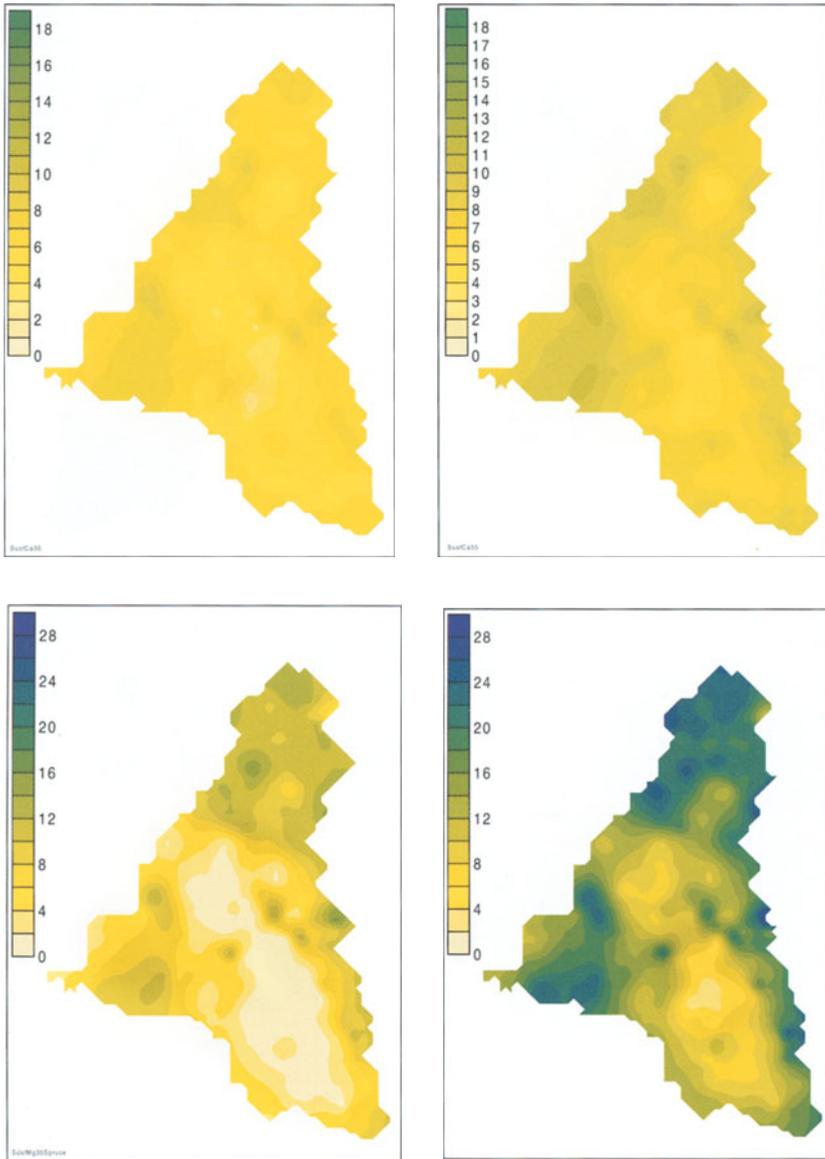


Figure 14.15: Sustainable yield in $m^3 ha^{-1} yr^{-1}$ based on Calcium for Norway Spruce in mono-culture (left, top) and a mixed stand (right, top). Below is the same for Magnesium, Norway spruce (left) and mixed stand (right). The sustainable yield is based on the assumption that Norway spruce has an average rooting depth of 0.35 meter. Sustainable yield in $m^3 ha^{-1} yr^{-1}$.

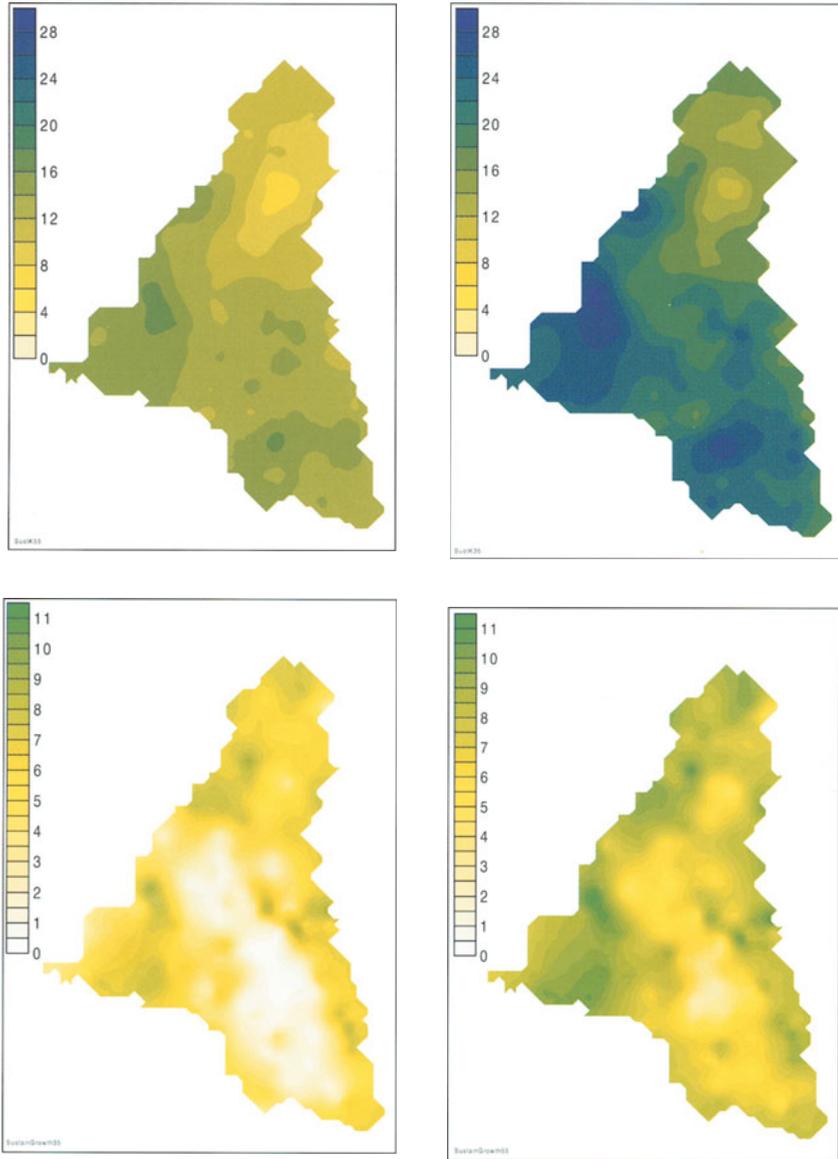


Figure 14.16: Sustainable yield in $m^3 ha^{-1} yr^{-1}$ based on Potassium for Norway Spruce in monoculture (top, left) and in a mixed stand (top, right). The two bottom maps show sustainable harvest based on the minimum allowed based on all base cations, Norway spruce monoculture (left) and to the right a 50:50 mixture of Norway spruce and Birch/Beech/Oak.

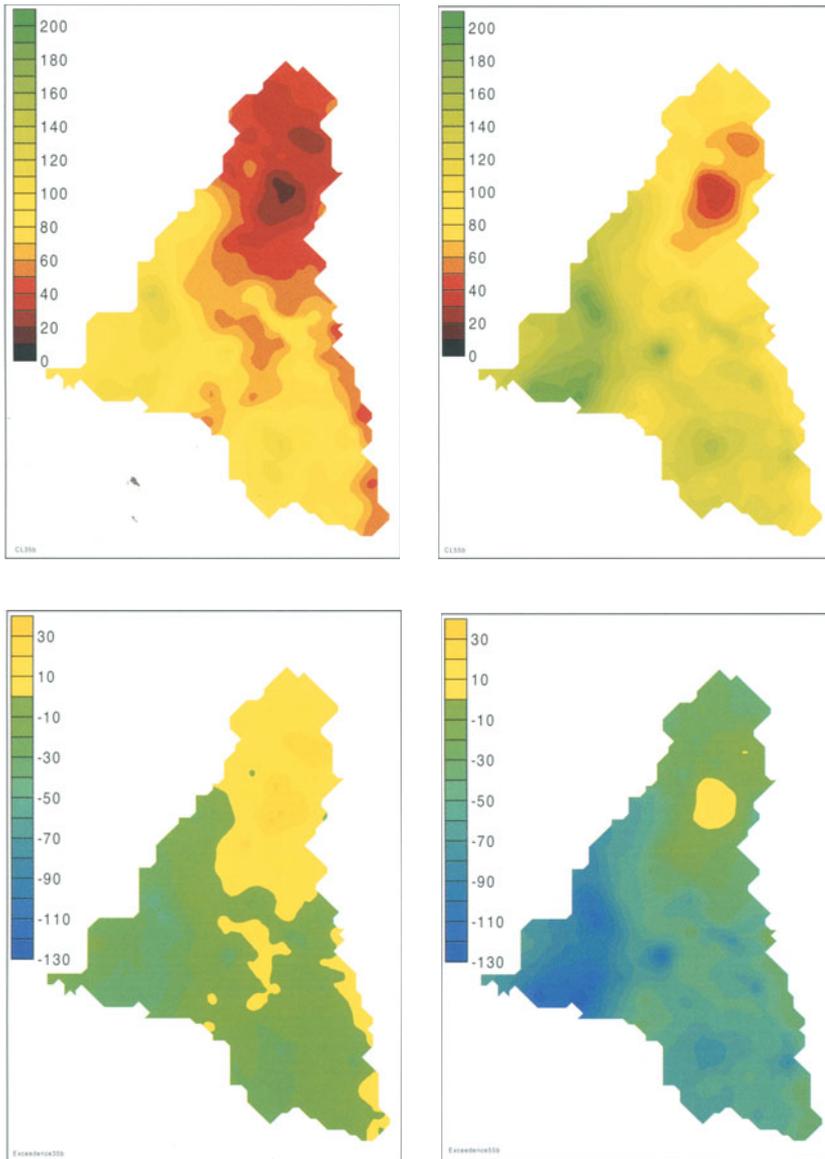


Figure 14.17: Critical load of acidity for conifer monoculture (top, left) and mixed forests (top, right). At all times in the future, the BC/Al ratio is predicted to stay well above the critical value of 1.0. Exceedance of critical loads of acidity for conifer monoculture (bottom, left) and mixed forests (bottom, right).

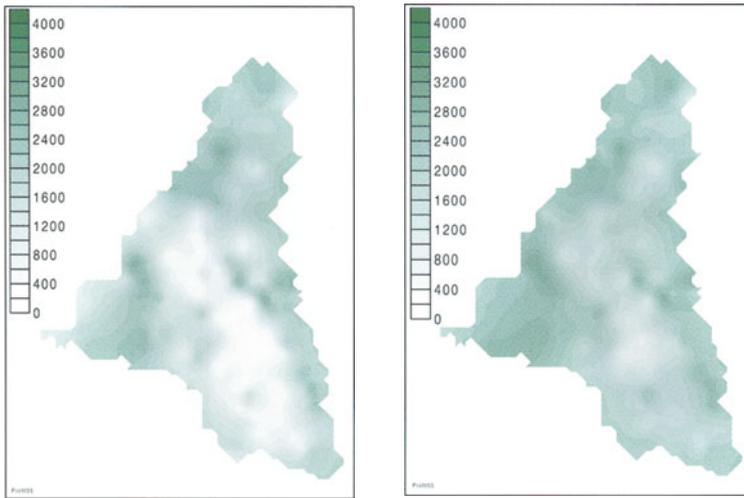


Figure 14.18: *Estimated areal distribution of net profits within the sustainability limits. On the left is the sustainable profit obtainable with Norway spruce monoculture, whereas the map to the right show the sustainable profit with a deep-rooted tree species, for example oak or beech. Such trees have potential for setting their roots at greater soil depth, but it must be remembered that this is a potential, and that deeper rooting depends on many factors such as physical and chemical site conditions and possibly on stand management. The higher profit figures can only be achieved in certain parts of the Asa Forest Park with favourable conditions and management. Units are SEK ha⁻¹yr⁻¹.*

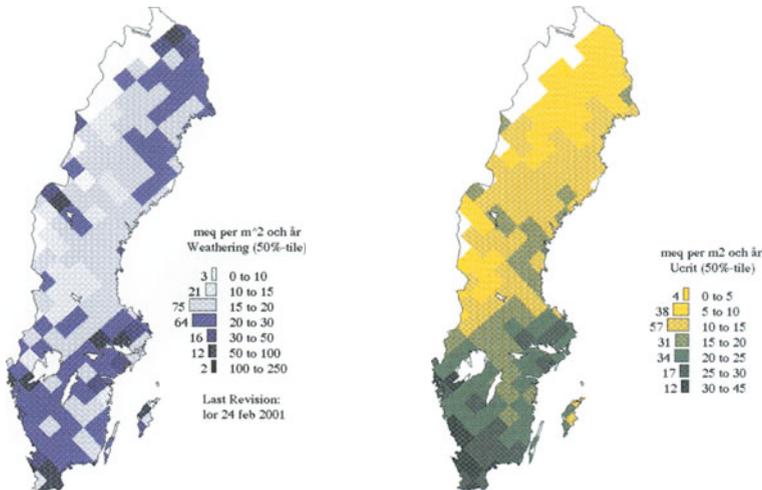


Figure 14.19: *National assessment of nutrient supply described in Chapter 15. Total base cation release caused by weathering of soil minerals was calculated using the PROFILE model is shown on the left. The sum of base cation weathering is shown, using present species distribution. The other map to the right shows the critical uptake of base cations available to the tree, as the minimum of the available amount of Ca, Mg and K.*

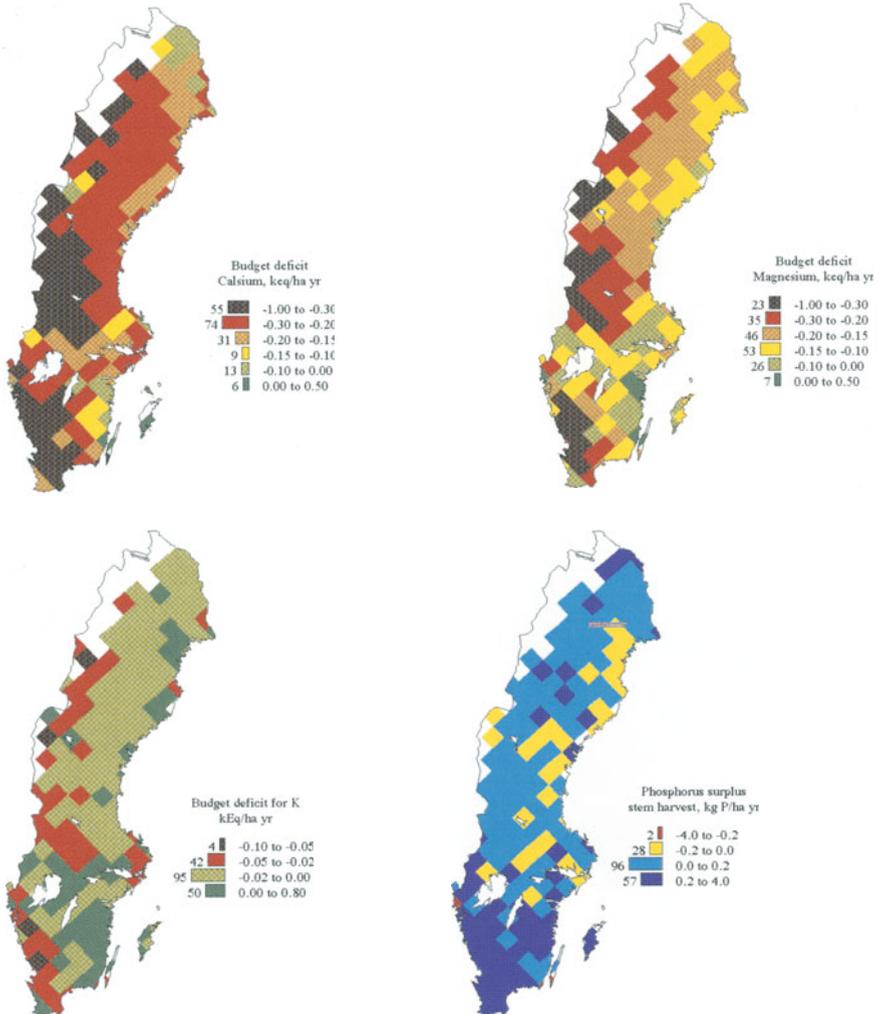


Figure 14.20: National assessment described in Chapter 15. Calcium (top, left), magnesium (top, right), potassium (bottom, left) and phosphorus (bottom, right) differences between supply and removal in Swedish forested stands, using year 1988 deposition and year 1990 leaching.

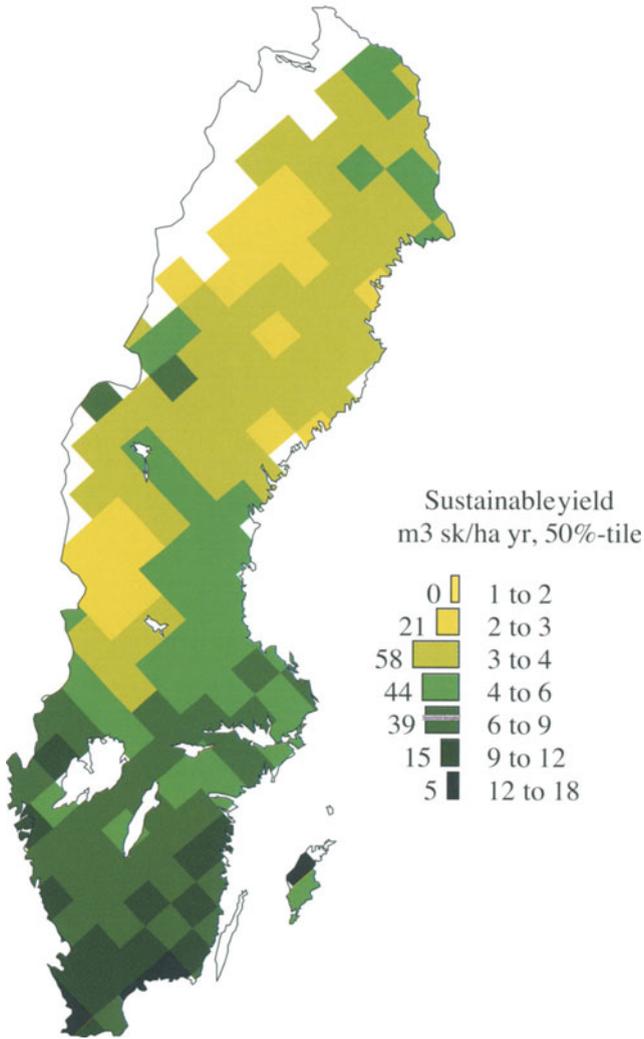


Figure 14.21: Sustainable growth for harvest in Swedish forests as described in Chapter 15, based on nutrients from weathering and base cation deposition alone. The map shows the 50%-tile, corresponding to average growth. In the SUFOR programme, an estimate of sustainability was one of the goals. This has been reached, we have achieved the capacity to set quantitative sustainability limits based on multiple criteria for multiple production goals set by the natural, social and economic spheres, for scales ranging from plots of a few hectares to large regions of hundreds of thousands square kilometres. This is an important milestone reached by the SUFOR Programme.

| Forest type | Spruce | Birch | Oak/Lime | Sum |
|---|--------|-------|----------|-----|
| Sustainable production volumes | | | | |
| $\text{m}^3\text{ha}^{-1}\text{yr}^{-1}$ | | | | |
| Spruce monoculture | 4.8 | - | - | |
| Spruce-birch mixed | 3.4 | 2.5 | - | |
| Mixed broadleaf | - | - | 5.2 | |
| $\text{m}^3\text{SPEha}^{-1}\text{yr}^{-1}$ | | | | |
| Spruce monoculture | 4.8 | - | - | 4.8 |
| Spruce-birch mixed | 3.38 | 3.72 | - | 7.2 |
| Mixed broadleaf | - | - | 9.3 | 9.3 |
| Realizable sustainable production volumes | | | | |
| $\text{m}^3\text{ha}^{-1}\text{yr}^{-1}$ | | | | |
| Spruce monoculture | 4.6 | - | 0.2 | |
| Spruce-birch mixed | 3.4 | 2.3 | 0.2 | |
| Mixed broadleaf | - | 0.4 | 3.1 | |
| $\text{m}^3\text{SPEha}^{-1}\text{yr}^{-1}$ | | | | |
| Spruce monoculture | 4.6 | - | 0.3 | 4.9 |
| Spruce-birch mixed | 3.4 | 3.5 | 0.3 | 7.2 |
| Mixed broadleaf | - | 0.6 | 5.5 | 6.1 |

Table 14.14: Summary of the production calculation expressed as real physical m^3 and Spruce Equivalent Units (SPE). At Asa Forest Research Park, broadleaf forest can only be realized at the best soils and in the best positions around the lake, maybe 10-25% of the total park area.

Sustainable yield

The sustainable yield is defined as the maximum yield that can be created based on the available nutrients in the system. In this first estimate, sustainability with respect to Ca, Mg and K will be addressed. The equation used was:

$$Y_{BC} = \min_i \left(\frac{BC_{U,i}^{crit} \cdot E_i}{\rho \cdot x_i} \right) \quad (14.23)$$

where $BC_{U,i}^{crit}$ is the critical uptake of element i =Ca, Mg, K, E_i is the equivalent weight of element i , ρ is the specific density of the harvested biomass and x_i is the content in the harvested biomass of element i . For Asa this was minimized over Ca, Mg, K. In most cases Mg was the element setting limitations, but for some cases also Ca was limiting. The critical uptake is limited by the available nutrient;

$$BC_{U,i}^{crit} = W_i + D_i - L_{min} \quad (14.24)$$

where W_i is the weathering release of element i , D_i is the atmospheric deposition of element i and L_{min} the minimum leaching from the system. For nitrogen the sustainability level is:

$$Y_N = \frac{N_U^{crit} \cdot E_N}{\rho \cdot x_N} \quad (14.25)$$

The critical uptake is given by what is long term available:

$$N_U^{crit} = 0.6 \cdot D_N \tag{14.26}$$

The total sustainable yield considering both base cations and nitrogen is:

$$Y = \min(Y_{BC}, Y_N) \tag{14.27}$$

This was estimated for Asa Forest Research Park. The sustainability map for production is based on available base cations in the soil, converted to stemwood biomass, if distributed among different tree species also converted to SPE. If whole tree harvest is practiced within the sustainability limit, wood production efficiency will be substantially reduced. Sustainable yield assuming a larger effective rooting depth is substantially larger than in a Norway spruce stand. The results of the sustainable production estimates have been shown in Figs. 14.16 and 14.15. There the maximum production based on each individual cation has been made. At present the corresponding limitation map was not made for phosphorus and nitrogen, but it would involve similar steps. The total maximum sustainable production is found by taking the minimum of these maps. Summary of the production calculation expressed as real physical m^3 and Spruce Equivalent Units (SPE) has been displayed in Tab. 14.14. For spruce monoculture and conifer-birch mixed forest, the sustainability potential can always be realized, for broadleaf forest, the slow growth of broadleaves set limits for how much of the sustainability potential that can be realized. At Asa Forest Research Park, broadleaf forest can only be realized at the best soils and in the best positions around the lake, maybe 10-25% of the total park area. Thus in a real management plan, all the components conifer monoculture, conifer-birch mixed forest, spruce-oak mixed forestry and mixed broadleaf should occur. For making comparisons easier, all volumes have also been converted to SPE.

This is shown in Fig. 14.22 where we also have made a comparison with the site yield map estimated according to the traditional method. It can be seen that even if the maps have been estimated with totally different data, the two maps qualitatively pick up the same pattern. The site yield maps has its high and low values in the same places as the sustainability map, but the site yield map is consistently higher. A conclusion must be that the site yield map is a qualitative map which is uncalibrated, where as the sustainability map is a quantitative map. One other way to see it is that the site yield map (Bonitet) illustrate the maximum growth management can make the system yield, the sustainability map show how much of the production speed it is reasonable to use.

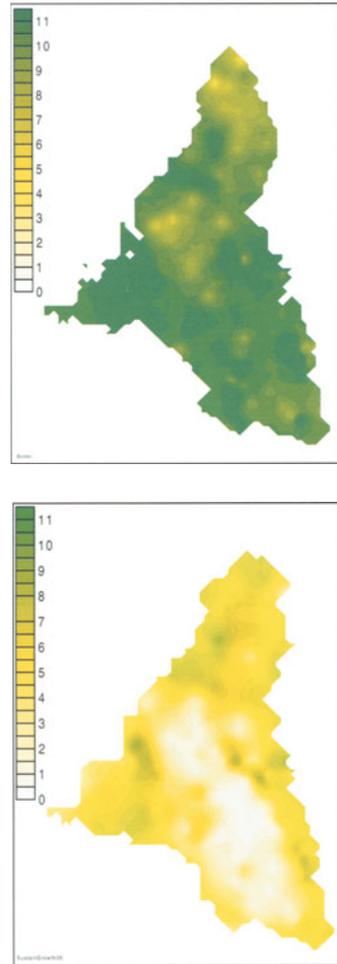


Figure 14.22: *Distribution of traditional forest yield over the registered stands at Asa is shown in the top map, assuming Norway spruce monoculture. This is to be compared with the corresponding map below, showing the sustainable yield as estimated in this study based on available K, Ca and Mg for the same Norway spruce monoculture. It is evident that the forest site yield (Bonitet) has been overestimated as compared to the sustainable yield. Units are $m^3 ha^{-1} yr^{-1}$*



Figure 14.23: A forest stream in southern Sweden. A reduction of the amount conifers in the immediate vicinity of the streams appears to improve the water quality. Photograph by Peter Schlyter.



Figure 14.24: Beech stands are important broadleaf trees that occur in substantial amounts in southern Sweden. Well managed, they will yield good profits on fertile soils in warm spots in the landscape, sometimes substantially better profits than obtained with the traditional management based on Norway spruce. Photograph by Peter Schlyter.

14.2 Sustainable economic profits

14.2.1 Introduction

For forest properties such as the one at Asa Research Park, economic viability is a necessity. The park is operated by employed personnel, who are paid from the proceeds of the operations of the Park. Although on a family farm property the labour costs would be less and the productivity possibly higher, a net profit is required in both cases. For many farms, this provides the proceeds needed for paying the inheritance tax at each generation shift and helps to obtain financing of the equipment for farmland management operations. It is thus important to investigate, the profitability of different management options. The calculations shown in this chapter is only a preliminary estimation. The principles applied are those described earlier chapters as well as the general principles of traditional forestry (de Jong et al 1999, Sillerström 1985)

14.2.2 Economic model

The model employed is described in earlier sections. The production is integrated forward and being expressed in physical production units, Spruce Pulpwood Equivalents (SPE), the costs of management such as for thinning, fencing, planting, soil tillage, etc., all being converted to SPE. The integral is divided by the rotation period, assuming that a given production volume is of equal interest regardless of the year of which it arises, thus the equal distribution to all years. The capital costs are balanced against production, eliminating interest on capital and using a straight investment payback over the rotation period. The profit P is calculated as the cumulative production minus all cost items, each converted into the same unit that of SPE. The productivity of a forest property can be measured in SPEP (Spruce Pulpwood Equivalent Productivity), expressed as $\text{m}^3 \text{ SPE yr}^{-1}$, the sustainability of productivity being expressed as SSPEP (Sustainable Spruce Pulpwood Equivalent Productivity), expressed as $\text{m}^3 \text{ SPE yr}^{-1}$. The profit for a forest plot in SEK is:

$$P = \sum_{j=1}^{j=m} A_j \left(\frac{350}{SPEEC} \cdot \left(\frac{1}{\tau} \cdot \int_0^{\tau} p \cdot dt - \sum_{i=1}^n C_i \right) \right) \quad (14.28)$$

where; A_j is the area of forest plot j from $j=1$ to $j=m$ (the last plot). SPEEC is Spruce Pulp Equivalent Cost (Price of wood divided by price of spruce pulpwood), SPE is Spruce Pulp Equivalents, giving the SPE of the wood produced in terms of $\text{m}^3 \text{ SPE ha}^{-1}\text{yr}^{-1}$, C is management costs in $\text{SEKha}^{-1}\text{yr}^{-1}$ of the n -th effort i , p is the production of wood in $\text{m}^3\text{ha}^{-1}\text{yr}^{-1}$ and τ is the time from one harvest to the next.

14.2.3 Results

Sustainability assessments of the base cations were used to estimate the different sustainable management scenarios. These scenarios represent components that can be used in preparing an integrated management plan in which other matters such as nitrogen management, carbon sequestration and biodiversity are also taken into account. The maps constructed show the profit potential under the condition that the sustainability potential can be fully realized in terms of actual growth. Site conditions may occasionally prevent this, for broad-leaves limitations of this kind can be substantial in certain parts of the area. At the same time, maps of how much production management can achieve for each of the tree species without soil nutrient limitations being exceeded need to be produced. Such maps indicating the maximum production and profit that can be achieved in the field, can be overlaid with maps of growth and of profits based on the sustainability potential so as to determine where Norway spruce, mixed spruce-birch forest and broad-leaved forest, respectively, would be most profitable. For Asa Research Park, it is evident that having a Norway spruce monoculture covering the entire area would not be the

| Trees species | Thinning % | Pulp wood % | B-Timber % | A-Timber % | Rotation time years |
|---------------------|---------------|----------------|---------------|---------------|------------------------|
| Norway spruce | 30 | 70 | - | - | 70 |
| Scots pine | 30 | 70 | - | - | 70 |
| Birch | 30 | 70 | - | - | 60 |
| Oak, lime and beech | 30 | 50 | 18 | 2 | 120 |

Table 14.15: *Rotation time and outtake of biomass for different silvicultural activities*

| Forest management | Rotation years | Harvest SPE m ³ | Clearance SPE m ³ | Planting SPE m ³ | Yield m ³ ha ⁻¹ yr ⁻¹ SPE |
|----------------------------|-------------------|-------------------------------|---------------------------------|--------------------------------|--|
| Spruce monoculture | 70 | 332 | 228 | 100 | 0.06-1.48 |
| Spruce and birch | 70 | 461 | 285 | 100 | 1.09-2.48 |
| Oak, lime and beech forest | 120 | 921 | 285 | 257 | 3.15-5.31 |

Table 14.16: *Product structure assumed for production at the Asa Research Park and calculations of production profitability using Spruce Pulpwood Equivalents (SPE) as the currency.*

most profitable management regime, assuming that growth needs to be sustained by the natural nutrient supply of base cations and of phosphorus. Because of the deeper root systems in a mixed forest and in a broad-leaved forest, such forests would have a higher sustainable yield than a monoculture of conifers would. For Asa the calculations indicate a landscape in which all three types of forest are found to be best, specially that a Norway spruce monoculture be located on only a relatively small portion of the area, and that there should be large swaths of mixed deciduous-conifer forest and a substantial fraction of broad-leaved forest in the warmer and fine-textured but well-drained soils as well as along the entire lake shore. In a system with a resupply of nutrients, one would need to add 1,000 kg of CaMg((CO₃)₂), corresponding to 2.5-3.0 ton ha⁻¹ of commercial grade (80% pure, efficiency 50%) dolomite, at each harvest. This would basically eliminate the differences in terms of sustainability potential between the tree species involved, so that the choice would depend on other factors, such as climatic factors, soil chemical conditions, water access, nitrogen and phosphorus in the soil, biodiversity, and the management intensity permitted. During a rotation period, this amounts to 150 kr/ha yr on the average. Although the range is 100-300 kr/ha yr, an additional dose of 100-300 kr/ha yr could be needed in order to restore severely acidified soils after pollution damages.

14.2.4 Conclusions

Under conditions of natural sustainability, a mixed conifer-deciduous forest or a pure broad-leaved forest is by a wide margin to be the most profitable type of forest for the Asa Research Park. It is ironic in a sense, that this was the makeup of the original forest before humans artificially replaced it by conifers. However, it is only possible to achieve an effective growth rate, one approaching the sustainability limit for broadleaves, on parts of the area, at low elevations around the shores of the lake, having a warm microclimate and covered mostly by

| Forest management | Natural nutrient supply and natural regeneration kr/ha yr | Natural nutrient supply and planting kr/ha yr | Natural nutrient supply, planting, fencing kr/ha yr |
|----------------------------|--|--|--|
| Conifer monoculture | 518 | 20 | - |
| Spruce and birch | 880 | 380 | -118 |
| Oak, lime and beech forest | 1,855 | 1,105 | 813 |

Table 14.17: *Costs in SEK for regeneration method and nutrient supply to different types of forest stands.*

| Forest management | Standard production, natl. repro., nutrient replacement kr/ha yr | Standard production, planting, nutrient replacement kr/ha yr | Quality wood, planting, nutrient replacement kr/ha yr |
|----------------------------|---|---|--|
| Conifer monoculture | 2,175 | 1,675 | 1,750 |
| Spruce and birch | 1,780 | 1,280 | 1,310 |
| Oak, lime and beech forest | 1,885 | 1,565 | 2,150 |

Table 14.18: *Estimation of profitability, expressed as annual profit per hectare. Profitability uses the projection of SPE over time for the elimination of interest rate, capital cost in excess of capital repayment thus not being included.*

brown soils. Thus, the optimum appear to be a mosaic that exploits the local properties of each stand or collection of stands. At higher locations and locations with poorer soil, the use of mixtures is especially important so as to enhance growth and increase the sustainability limit for growth. Whole-tree harvesting is not sustainable in the park under any circumstances, the fertility of the soil is rapidly eroded by such practices. In a management regime involving nutrient management and replacement at every harvest, higher production can be sustainably maintained. Whole-tree harvesting anywhere in Sweden requires nutrient recycling practices, so as to not ruin the soil and leave it infertile and depleted. The differences in profit between the various tree species that can be selected are not very large, although a spruce monoculture is advantageous if pulpwood is the product targeted (2,175 SEK ha⁻¹yr⁻¹). If wood of high quality targeted, broad-leaved forest or sequential conifer-broadleaves mixtures would be best whenever the local conditions enable the growth rate to be reasonably high (2,150 SEK ha⁻¹yr⁻¹). In a purely deciduous landscape, broad-leaved forestry for producing quality wood would be superior in profitability to other forms whenever the local conditions allow a reasonably high growth rate to be achieved (2,900 SEK ha⁻¹yr⁻¹). In most cases, some degree of mixing with deeper-rooted species has the potential of being just as profitable as a conifer monoculture but will tend to be more sustainable.

| Forest management | Natural regeneration, fencing for deciduous SEK ha ⁻¹ | Artificial regeneration, fencing for deciduous SEK ha ⁻¹ | Deciduous landscape, no fencing SEK ha ⁻¹ |
|----------------------------|--|---|--|
| Spruce monoculture | 80,000 | 170,000 | 80,000 |
| Spruce and birch | 190,000 | 280,000 | 100,000 |
| Oak, lime and beech forest | 210,000 | 300,000 | 120,000 |

Table 14.19: Estimates of the approximate capital investment needed for establishment of a new mature stand starting from clear land.

14.3 Approximate carbon balances

14.3.1 Introduction

Forestry has a profound effect on the carbon cycle of a forest ecosystem, both for the soil and for the forest stand in general. A forest not only has a strong capacity for sequestering large amounts of carbon dioxide, but can also emit quantities of the same magnitude. Although an undisturbed ecosystem normally shows a small net gain in carbon annually, the carbon exchange with the atmosphere in a managed forest stand can sometimes be negative (Waring and Running 1998). There are two major carbon sinks in a forest, the vegetation itself and the soil organic matter. the latter being the most important long-term carbon sink. The organic matter in a forest ecosystem remains unoxidized for centuries, carbon accumulating in the forest floor (Swift, Heal and Anderson 1979). If the soil organic matter layer is disturbed, strong emissions of carbon can take place (Kimmins 1997). Disturbances in the vegetation layer caused by human management such as thinning, harvesting and natural disturbances such as wild fires and storm felling, can suddenly result in a reduction in photosynthesis, so that the ecosystem loses carbon to the atmosphere. In situations of this sort, the net ecosystem production (NEP) is negative. The NEP in a managed forest stand is mainly the result of three different factors: forest management strategies, harvesting strategies and land-use history. An approximate measure of the carbon balance in a managed forest can be obtained by use of simple analytical tools and methods. In the present paper, this is exemplified for the Asa estate area in the province of Småland in southern Sweden.

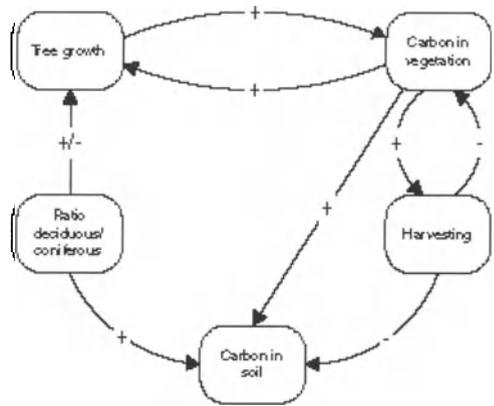


Figure 14.25: Schematic presentation (causal loop diagram) of the carbon cycle in a forest ecosystem, adjusted for long-term carbon budget studies.

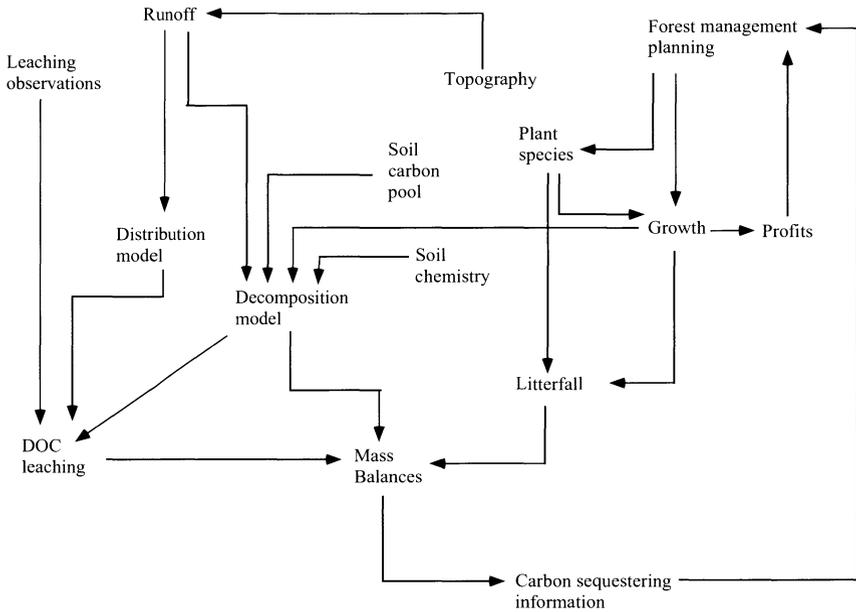


Figure 14.26: Flow diagram for assessment of carbon sustainability in forest management. The assessment procedure required close cooperation between the different programme parts.

14.3.2 Methods

The methods employed in the study are those of simple "back-of-an-envelope"-type calculations. The input data consist of field measurements that, together with various estimations and assumptions, are used in mental models and standard mass balances. The results can be evaluated from the standpoint both from present knowledge and of reasonableness. A mass balance is merely a bookkeeping system that keeps track of the sources and sinks within the boundaries of the system.

$$IN + PROD = OUT + ACC \tag{14.29}$$

That which goes into the system plus that which the system produces must equal that which goes out plus that which is accumulated. What the different terms in the mass balance equation stand for depends on the time-scale employed. On an hourly or daily basis, the *IN*-term represents carbon assimilation, the *PROD*-term photosynthesis, the *OUT*-term respiration and the *ACC*-term allocation. In a long-term study extending over decades or centuries fluctuations in photosynthesis and in respiration are unimportant. The *PROD*-term in such a case is annual growth, the *OUT*-term removal of the crop and of soil and the flux of dissolved organic carbon (DOC) together with water and the *ACC*-term carbon sequestration in the standing crop or in the litter on the forest floor. The *IN*-term is only useful if one practices ash restoration or other means of putting carbon into the forest manually. The *PROD*-term is calculated using the formula

$$PROD = \sum_{i=startyear}^{i=endyear} growthrate_i \cdot area_i \tag{14.30}$$

To estimate the growth rate term, a thousand-year period was divided into five shorter periods, see Figure 14.27. The tree-growth rates were considered to be linear during these periods and

to be twice the stem-growth rates. If one starts with the stem-growth rate known to exist today, one can assume that the next 100 years will have approximately the same growth rate. Beyond year 2100, however the decrease in nitrogen deposition can be expected to become noticeable and the growth rate to start to slowly decrease. The period of 1900-2000 which has just gone by was characterized by a rapid increase in nitrogen deposition and a gradual increase in the efficiency of forest management, and thus by, a rapid increase in rate of growth. Back in the late 1700s and 1800s, despite forests being managed in an increasingly efficient way, no nitrogen effect occurred, Thus, the increase in growth rate then was less than in the 1900s. For the period of 1250 to 1750, the growth rate is considered to only have increased as a result natural causes.

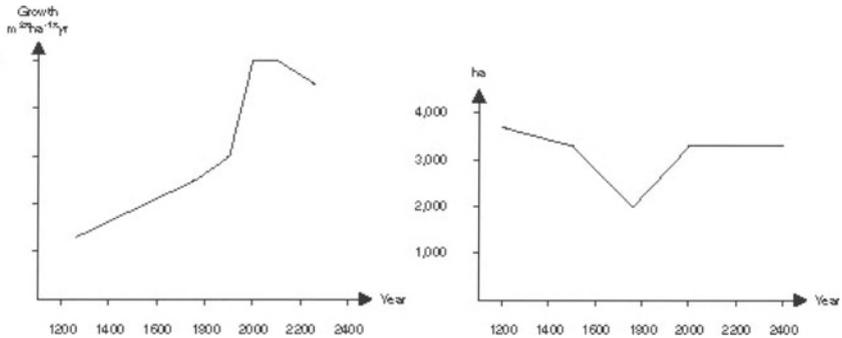


Figure 14.27: *Left: The estimated stem-growth rate at Asa. The rate shows an increase, since management became more efficient during the latter half of the 1700s. Nitrogen deposition caused by industrialization boosted the growth rate in the 20th century in combination with still more efficient management methods. Right: Assumed changes in the size of the forested area at the Asa estate.*

The present account of the development of forested area is based on historical sources (Sjöstrand 1952, Andersson 1996). The period of 1250 to 2250 AD was divided into four different sub-periods such that the developments during each could be described by a set of linear equations, see Fig. 14.27. The *OUT*-term represents what is taken out of the forest ecosystem. In the present case, two major terms are focused upon: harvesting and loss of carbon due to drainage actions. In the first period considered, harvesting was virtually null. *OUT* during the next period is regarded being equal to 10% of the annual stem growth, whereas for the last two periods it is regarded being 20%. Drainage measures are considered to have resulted in the emission of half of the carbon accumulated in the soil. For all four periods, the carbon that accumulates in the soil is assumed to represent 10% of the total annual growth.

14.3.3 Input data

The data were taken mainly from two sources, recent investigations of the Asa estate together with historical accounts of it (Sjöstrand 1952, Andersson 1996), and the Swedish National Forest Inventory (Swedish University of Agricultural science 2000). Swedish forests have a history of continuous changes, Asa being no exception. The forest-covered area has changed over time. It was smallest at around 1850, being smaller then, in fact than at any time in the past 7,000 years. A large part of the landscape was pasture land in which there were occasional scattered groups of trees, a significantly smaller part of the area being dense forest. Since around 1870, the forest has gradually been taking over the landscape. Since the middle of the 19th century the forested area has increased by more than 50%.

During this period, the mixture of tree species in the Asa forests has changed considerably and the tree density has doubled due to replantations. Around 1750, the forests in Asa consisted to about 70% of deciduous trees and to 30% of conifers (Andersson 1996). Further back in time, some 1,000-2,000 years ago, the fractions of deciduous and of coniferous trees were more or less constant, about 90% being deciduous and 10% coniferous, see figure 14.29. Today the composition is one of about 25% deciduous and 75% coniferous trees. Pollen analysis shows that the impact of the humans on the species composition in Asa began around 750 years ago (Andersson 1996). In the first half of the 19th century, the population of the Asa village also increased rapidly, see Figure 14.28. It is reasonable to think that the agricultural activities showed a similar pattern of increase, the area of pastures and fields increasing at the expense of forested land. A steam-driven sawmill was built in Asa

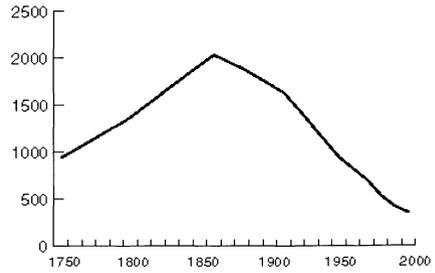


Figure 14.28: Changes in the population at Asa during the period 1750-1990. Around 1860, the large emigration to America began in this part of Småland. This pattern is assumed to also be reflected in the development of cultivated land in the area.

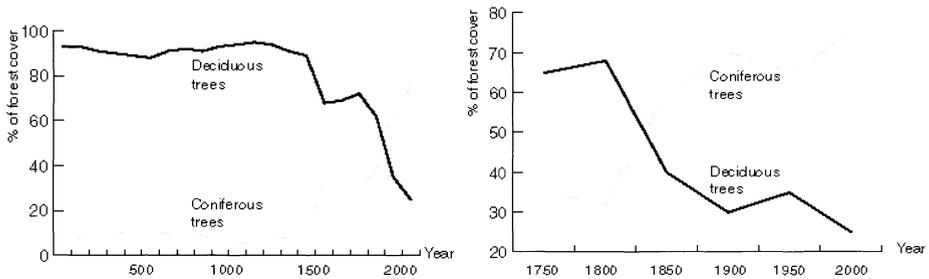


Figure 14.29: Right: Changes in the tree species composition at Asa during the last 2000 years. Left: The covering of broad-leaved trees increased on the cut areas at the beginning of the 20th century but were thinned out in accordance with modern forestry practices.

in 1866 (Sjöstrand 1952) and it was expected then that the vast areas of mature timber forest, considered the most beautiful and valuable in the entire province of Småland, would be clear-cut. Although it never went that far, at times the harvesting was very intense. In 1895 the standing timber was estimated to amount to 12,500,000 cubic feet, about 340,000 m³ or 125 m³·ha⁻¹ (the estate comprised 2,600 ha at that time), as compared with 146 m³·ha⁻¹ today. The harvest during the last few years of the 19th century was stated to have been 60,000 logs per year, almost a sixth of the standing timber. The replantation during those years amounted to some 80,000 coniferous trees, besides the sowing of seeds (Sjöstrand 1952). For a time at the beginning of the 20th century, the broadleaves took over a larger area again, but modern forestry put an end to that. The increase in harvesting together with the changes in tree species composition created by the one-sided replantation changed the carbon balance in the forest ecosystem considerably.

14.3.4 Calculations and results

In the typical coniferous forests in the Asa area, the soil can be said to consist of three different horizons. The litter layer on the forest floor is 1 cm thick and the organic layer 5 cm thick these

having an average carbon content of 32 weight-% DW and an average bulk density of $230 \text{ kg} \cdot \text{m}^{-3}$ (Johan Holmqvist, unpublished data). This adds up to a carbon content of the organic layer of about $37 \text{ tons} \cdot \text{ha}^{-1}$. The average B-horizon in Asa is about 40 cm in depth, with a bulk density of $1070 \text{ kg} \cdot \text{m}^{-3}$ and a carbon content of 2.1% (Johan Holmqvist, unpublished data). This portion of the soil amounts to $90 \text{ tons} \cdot \text{ha}^{-1}$. The C-horizon is assumed to contain carbon down to a depth of 80 cm, the maximum rooting depth for Norway spruce, see Tab. 14.20. The bulk density of it is $1400 \text{ kg} \cdot \text{m}^{-3}$ and the average carbon content 0.67% (Johan Holmqvist, unpublished data). The C-horizon thus contributes about $30 \text{ tons} \cdot \text{ha}^{-1}$ of carbon. Hence, the total carbon content of the coniferous forest is $157 \text{ tons} \cdot \text{ha}^{-1}$. For deciduous forests in southern Sweden we use figures taken from the Swedish Forest Inventory (Riksskogstaxeringen), since the field data from Asa that are available are insufficient. The top layer consists of about 25 cm of humus organic matter containing approximately $68 \text{ tons} \cdot \text{ha}^{-1}$ of carbon. The typical brown earth horizon found in deciduous forests in Småland is 40 cm thick and contains 3.5 weight-% carbon, or about $180 \text{ tons} \cdot \text{ha}^{-1}$. The carbon containing part of the C-horizon is considered to extend down to a depth of 100 cm and to contain 2.4% carbon or $84 \text{ tons} \cdot \text{ha}^{-1}$. This represents a total carbon content of the meter-deep soil column of around $332 \text{ tons} \cdot \text{ha}^{-1}$ (Swedish University of Agricultural science 2001). Figure 14.30 presents the calculated carbon content of the horizons for the different soil types. One reason for these differences is the difference in root distribution between deciduous forest and a Norway spruce forest, see Table 14.20, (Kimmins 1997, Swift *et al.* 1979). Deciduous trees put more carbon into the deeper soil horizons than Norway spruce do, almost all of the roots of which are in the upper 10 cm of the soil. For the agricultural land in the area, an average carbon content of 4 weight-% appears reasonable (Eriksson 1990, Eriksson, Andersson and Andersson 1997), the profile being assumed to be one of the topmost 25 cm of having a carbon content of $100 \text{ tons} \cdot \text{ha}^{-1}$ and the 50 cm layer beneath containing $3 \text{ tons} \cdot \text{ha}^{-1}$ of carbon. Note that the differentiation of the soil types referred to, does not appear to be based on the historic use of the land, there being no difference in this respect between an area of spruce growing on old arable land and an area of it growing on an old deciduous forest site. Today, there is a productive forest area on the Asa

| Soil depth (m) | Coniferous (weight-%) | Deciduous (weight-%) |
|-------------------|--------------------------|-------------------------|
| 0-0.20 | 80 | 55-65 |
| 0.20-0.55 | 20 | 25-35 |
| 0.55-0.80 | - | 5-10 |

Table 14.20: *Approximate root distribution in weight-% in different soil layers for different types of forests.*

estate of 3,300 hectares, as compared with 2,600 hectares in 1895 (?). Since the Asa estate was already run as a large-scale agricultural business in the late 1600s, the forested land is estimated to have encompassed as little as 2,000 hectares in 1750, as shown in Figure 14.27. The exchange of carbon between the soil and the atmosphere has been influenced by the change in land use. A change from deciduous forest to agricultural land is the most expensive type of change from the standpoint of carbon balance, see Table 14.21. In ancient times, before the ecosystem had been affected by man, there was almost 1,200 kT of carbon altogether in the soil of the Asa area, whereas in 1750 the amount was 770 kT and today it is less than 700 kT. The estimated areas and changes in area, expressed as changes in soil carbon content for the different periods, are shown in Figure 14.31. Thus, the forest soils in Asa have lost some 475 kT as compared with the situation about 750 years ago, due to changes in the species composition. The positive side of this mass balance is the increase that has occurred in the total carbon stored in the standing

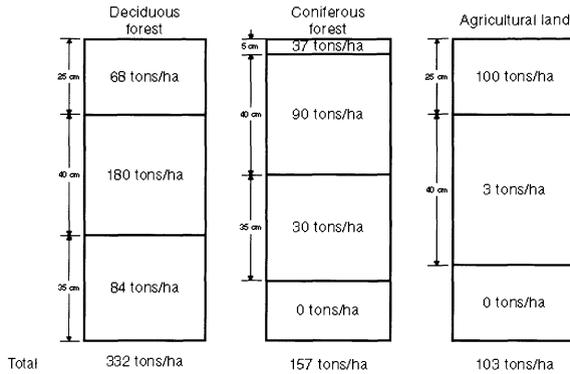


Figure 14.30: Carbon content in the different soil layers in deciduous and in coniferous forests and on agricultural land.

| Type of change | tons·ha ⁻¹ |
|--|-----------------------|
| Deciduous forest to agricultural land | -229 |
| Deciduous forest to coniferous forest | -175 |
| Agricultural land to coniferous forest | +229 |
| Coniferous forest to agricultural land | -54 |
| Coniferous forest to deciduous forest | +175 |

Table 14.21: Results of land-use changes in Asa Forest Research Park.

timber. Forest management and increased nitrogen deposition have increased the productivity of the forest, at the same time as the old agricultural land has been reforested. In 1250 The store of standing timber is estimated to have been 100,000 m³ in 1250, and 200,000 m³ in 1750. In the 1998 inventory it was estimated to amount to 480,000 m³. This represents a 190 kT increase in the carbon store during the period of 1250-2000. The enhancement of the primary production which has occurred is also reflected in the soil. Ten percent of the total annual turnover is assumed to be permanently immobilized. The growth rate has increased from what is assumed to have been 2.5 m³·ha⁻¹·year⁻¹ in 1250 to 10 m³·ha⁻¹·year⁻¹ in 2000, and it can be expected to slowly decline as the nitrogen deposition becomes less, see Figure 14.27. For the period of 1250-2000, this reasoning leads to a result of 250 kT of carbon having sequestered and accumulated in the soil. Half of the total carbon accumulated, or 260 kT, is considered to have been emitted during the period of 1750-2000 due to drainage measures. A certain amount of carbon is of course immobilized in wood products like books, houses, furniture or whatever. This part is estimated to represent 10% of the stem growth. The carbon balance is summarized in Table 14.22.

14.3.5 Discussion

The method of "back-of-an-envelope-calculations" is a fast and simple tool for making estimates concerning ecological systems. The magnitudes of the different variables, process rates and other quantities involved provide an idea of their relative importance to the ecosystem. In the present case, involving the carbon balance in the Asa estate, the most important factor is clearly that

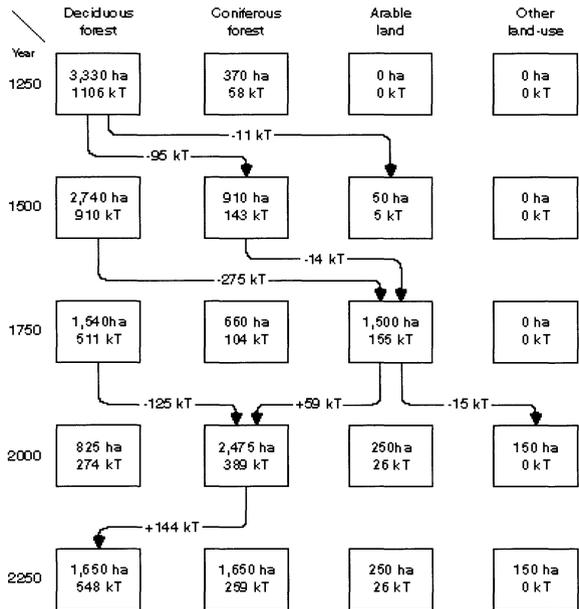


Figure 14.31: Estimated historical changes in area and in carbon relocation between stores due to changes in land use. Carbon flows (arrows) are in kT carbon (10⁶ kg).

of the tree species composition. Some 5,000 years ago, the predominately oak-beech forests found in southern Sweden, were cut . In 1840, before the mass exodus of emigrants to America, the forested area constituted only 60% of that found at present, or 16 million as compared with 27 million hectares. Today 95% of the land in southern Sweden is cultivated. The species composition and age distribution of the forests is the work of man. In southern Sweden some 50% of the spruce stands are 41-80 years old, meaning in this part of the country that that they are ready for cutting, or will be ready for cutting in less than half a rotation. Cutting such large amounts of forest will lead to large losses in carbon, particularly in views of the harvesting methods and strategies are employed. Perhaps as much as 25% of the total area of southern Sweden has been drained during the last 150 years. This has undoubtedly released large stores of carbon from waterlogged areas. A reasonable assumption is that 2,000 to 5,500 million tons of carbon were released during the period of 1870-1970. For the area of Götaland (the southernmost third of Sweden) it should take approximately 200-1,000 years to restore the soil-carbon stores to their former level. In Asa the situation is even worse, due to the often very heavy-handed forestry and draining procedures employed, see Fig. 14.32. It would take more than 2,000 years for the carbon stores there to be replenished. The permanent total carbon sequestration in Asa was estimated to be 10% of the sustainable yield. Although still larger amounts can accumulate during transitional periods, accumulation of this kind is not permanent. The harvesting methods employed determine to a large extent how much will be re-emitted into the atmosphere. Re-emission occurs then during a very short period of time. The expected recovery from soil acidification during the period of 2010-2100 is expected to reduce the permanent sequestration by 2-3%-units due to the conditions for decomposition processes improving.

| Carbon store | 1250 | 1500 | 1750 | 2000 | 2250 |
|-----------------|-------|-------|-------|-------|-------|
| In soil | 1,164 | 1,058 | 770 | 689 | 833 |
| Immobilized | 0 | 140 | 290 | 250 | 650 |
| Standing timber | 50 | 90 | 100 | 240 | 210 |
| Sum | 1,214 | 1,288 | 1,160 | 1,179 | 1,693 |
| In products | 0 | 80 | 300 | 700 | 1,100 |
| Sum | 1,214 | 1,368 | 1,460 | 1,879 | 2,793 |

Table 14.22: The carbon content of the forest ecosystem in Asa at different times in history. All numbers are in kT of carbon (10^6 kg C)

14.3.6 Conclusions

Carbon sequestration depends on two major factors, the supply of nitrogen and the management strategy employed. The largest permanent storage of carbon is in the soil, only smaller amounts being able to be stored in the aboveground biomass. Forestry involving mixed stands (both deciduous and coniferous trees) and stem-only harvests contribute positively to the sequestration of carbon. The nitrogen supply determines to a large extent the decomposition rate of organic matter. It can be concluded that a mixed forest builds up a larger carbon pool in the soil, but that this reaches a steady-state in approximately 50-100 years and that only modest amounts of carbon are permanently sequestered. The carbon stores contained in standing timber are of only very slight importance in the long run. Less than 10% of the carbon in a coniferous forest and less than 5% of that in a deciduous forest is contained in the vegetation.

Carbon sequestration in Swedish forests is a delicate process. Even small disturbances can reverse it. The soil is the most important carbon sink and should be managed with good care. It is important that harvesting and soil preparation prior to replantation be carried out in a way allowing both forests and topsoil to remain productive in the future.

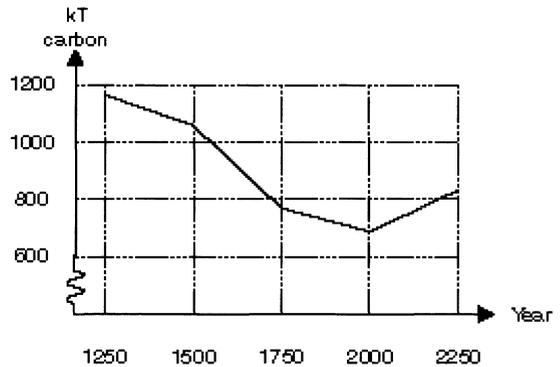


Figure 14.32: The carbon stores in the Asa soils began to change 750 years ago when the conifers appeared. The expansion of agriculture during the period of about 1600-1850, led to many forest areas being cleared. During the 1800s, many persons from this region emigrated to America and extensive cultivated areas were reforested.

Chapter 15

Nutrient sustainability for Swedish forests

Harald Sverdrup

15.1 Introduction

The work with critical loads for acid deposition, created the expression "critical uptake" as a term to be used in the calculation of nitrogen critical loads (Sverdrup et al. 1990; Warfvinge and Sverdrup 1992, Sverdrup and Rosen 1998). These "critical uptakes" are expressions of how much the forest can take up without depleting the soil. If the critical uptake is adjusted to management and inverted, the sustainable harvest can be estimated. Thus we can address if the forest at present runs a profit or a deficit with respect to base cations, but we can also ask the question of how much wood can we produce with the base cation supply available ? This is what we will do in the following.

15.2 Input data

All data in this work is based on data and soil samples collected in the Swedish Forest Inventory (Rikskogstaxeringen/Ståndortskarteringen). The inventory data was collected 1983-1985. It consists of a network of 1,884 plots evenly spread over the complete forest area of Sweden (Approximately 227,000 km² or 22,700,000 ha in 1987). Soil samples down to approximately 0.5 m depth were collected during 1983-87. The basic soil parameters for this study were analyzed on these samples. The list below represent the input data required for the PROFILE model for each site in the calculations; total deposition of sulfate, nitrate, ammonium and base cations, precipitation, runoff, temperature, soil mineralogy, soil texture class, base cation uptake, nitrogen uptake, canopy cycling of base cation and nitrogen, thickness of the soil layers, soil type and soil moisture saturation.

Absolute soil mineralogy was derived for 140 sites by measurement by the Swedish Geological Survey (SGU) at Uppsala and at the Czech Geological Survey in Praha. Total elemental content of the soil after complete dissolution in a lithium borate melt was determined for all 1884 sites. These were analyzed for Ca, Mg, Na, K, Al, Si, Fe, Ti and trace metals (Warfvinge and Sverdrup 1993). The UPPSALA model is a back-calculation model for reconstructing the mineralogy from the total digestion analysis in order to provide input to models like PROFILE from simple survey data (Sverdrup and Warfvinge 1991). The minerals have been grouped into assemblies of minerals with similar composition and dissolution rate. Texture was measured by granulometry and BET/adsorption analysis on the 107 mineralogy analysis samples, and correlated against field texture classification. The texture for all sites were read from the correlation using the field classification. A relation between the field classification of soil texture and laboratory

measurements of exposed surface was found:

$$A_{tot} = 0.093 \cdot e^{(0.51 \cdot TX)} \quad (15.1)$$

where TX is the texture class, A_{tot} is the total surface in $10^6 \text{m}^2 \text{m}^{-3}$ soil. The relation is based on determinations of surface area and texture class on 100 soil samples. Deposition data were prepared by the Institute of Water and Air Research (IVL) at Göteborg, from the Swedish deposition monitoring network and EMEP data. Modified with filtering factors for different vegetation types. The total deposition is calculated for each calculation point using the filtering factors and the vegetation mixture for that point. The deposition used correspond to 1988 deposition level. The Institute of Air and Water Research at Göteborg estimates the uncertainty in the base cation deposition values to be as much as $\pm 50\%$, causing this to be one of the largest single sources of uncertainty.

15.3 Results

Fig. 14.19 show the total weathering rate of Ca, Mg and K in the rooting zone, for Norway spruce 0.35 m, for pine 0.45 m, for birch 0.55 m and for broadleaves 0.65 m. Neither the weathering rate of Mg nor Ca follow the pattern of N deposition and present forest yield. This mismatch predisposes Swedish forests for nutrient imbalances induced by high N deposition. The average soil weathering rate for the rooting zone in Swedish soils have a median value of $0.22 \text{ keq ha}^{-1} \text{yr}^{-1}$, but only $0.18 \text{ keq ha}^{-1} \text{yr}^{-1}$ of this is useful as tree nutrition as Ca, Mg and K. This low average value is caused by a majority of the soils having minerals of granitic origin and the average coarse texture of Swedish forest soils. No single soil site in the database has a weathering rate value above $4.5 \text{ keq ha}^{-1} \text{yr}^{-1}$ in the rooting zone. The leaching rate for Mg as calculated from runoff and generalized soil solution concentration at approximately 0.5 m were calculated and calibrated against some few field observations. These are approximately 40 points, and not statistically representative in a strict sense. But they are the only information available. The maps show large negative differences between supply and removal for both Ca and Mg, but less for K. Fig. 14.20 show the results of the calculations. Negative implies that removal by growth and leaching is larger than supply, in the long term there will be too little. The diagram in Fig. ?? show the distribution of negative differences over all sites. The calculations show that approximately 95% of the area has net negative differences between supply and removal for Ca, 93% of the sites for Mg and approximately 67% of the sites for K. The average total base cation negative difference between supply and removal is $0.33 \text{ keq ha}^{-1} \text{yr}^{-1}$, equivalent to $180,000 \text{ ton yr}^{-1}$ of base cation carbonate or $0.39 \text{ ton ha}^{-1} \text{rotation}^{-1}$ of the missing ions expressed as carbonates.

If whole tree harvest is done without base cation return, then the negative difference between supply and removal for Ca will increase with approximately 50% and the negative difference between supply and removal in K even more. With whole tree harvest without base cation return, base cation negative difference will occur in 96% of the area. The average total base cation negative differences between supply and removal under whole tree harvest is $0.62 \text{ keq ha}^{-1} \text{yr}^{-1}$, twice the amount for stem harvest. In total, that implies an amount of base cations for the whole forested area of $320,000 \text{ ton yr}^{-1}$ if expressed as limestone, equal to $0.7 \text{ ton ha}^{-1} \text{rotation}^{-1}$. The result of the calculation is that whole tree harvest on the average double the base saturation depletion rate. With a depletion rate of $0.33 \text{ keq ha}^{-1} \text{yr}^{-1}$, it would take from 15 to 100 years to empty the present base cation store in the forest soils. The depletion rate varies from approximately from less than 1% to as much as 10% per year. If the depletion rate is increased to $0.62 \text{ keq ha}^{-1} \text{yr}^{-1}$, the depletion time is reduced to a range from 8 to 50 years (20-2% depletion per year), or in severe cases substantially less than a rotation period. There are obviously large uncertainties in such calculations, but still it illustrates that the problem must be taken seriously. A summary of some of the results of the calculations can be seen in Tab. 15.1-15.2. In addition to the case using present day values for the input parameters, a

best case was constructed. The best case assume that the trees do uptake so effectively that they drive the soil solution concentration down to $5 \mu\text{eq l}^{-1}$ for Ca and Mg and $1 \mu\text{eq l}^{-1}$ for K. A second case was constructed, whole tree harvest, which assumes consistent removal of the whole tree, stem, branches and needles upon harvest. Figure 14.20 and 14.20 shows one example of the results that are obtained when a budget calculation is made for Swedish forests. The example shows a budget balance for magnesium, the vital element for making chlorophyll. The map is based on data from the Swedish forest inventory. Such maps show large negative differences between supply and removal for both Ca and Mg, but less for K. Negative implies that removal by growth and leaching is larger than supply, in the long term there will be too little. The calculations show that approximately 95% of the area has net negative differences between supply and removal for Ca, 93% of the sites for Mg and approximately 67% of the sites for K in the period 1987-1993. The average total base cation negative difference between supply and removal is $0.33 \text{ keq ha}^{-1}\text{yr}^{-1}$, equivalent to $0.39 \text{ ton ha}^{-1}\text{rotation}^{-1}$ of the missing ions expressed as carbonates. From the perspective of sustaining high growth rates, problems may arise. The amount nitrogen available for tree uptake remains high, but the amount of Ca, Mg and K or all decrease. The plant requires relatively fixed ratios of nitrogen to individual base cations in order to make needles, bark, branches, stems and roots, and the uptake elasticity is not very large. If the base saturation would reduce to very low values, the soil solution would in many places have high concentrations of Al that potentially make problems for tree roots and the stability of the ecosystem would be low. The base saturation is important, as a source of short-term resources to cover short term high needs of base cations. Such "loans" can be repaid in other seasons with normal weathering but less growth.

15.4 Discussion

Every year since the acidity deposition at a certain site exceeded the weathering rate, the base saturation decreased with an amount close to the difference between the incoming acidity and the weathering rate. It goes almost without saying that the reservoir of exchangeable base cations should become very low after some time. When the base saturation reach very low levels, it becomes very difficult to remove additional base cations, and the concentration of base cations in the soil solution will decrease, because the total input of such ions to the solution is less. Many of the soils in question has a base saturation of 5-15% and a CEC in the range from 100 to 250 keq ha^{-1} . This implies that the exchangeable amount of base cation in the soil is in the range from 5 to 35 keq ha^{-1} . With a depletion rate of $0.33 \text{ keq ha}^{-1}\text{yr}^{-1}$ it would take from 15 to 100 years to empty the base cation store. If the depletion rate is increased to $0.62 \text{ keq ha}^{-1}\text{yr}^{-1}$, the depletion time is reduced to 8 to 50 years, or less than a rotation period. There is obviously large uncertainties in such calculations, still it illustrates that the problem must be taken seriously. From 1840 to 1980 the acid deposition increased steadily, from 1980 to 2000 it has decreased by approximately 60%. During the same period growth was increased by more efficient management. At the same time the deposition of nitrogen increased, and depletion of the base saturation by acidification caused the concentration of Ca, Mg and K in the soil solution to increase. Both nitrogen and Ca, Mg and K was available to sustain the increased growth. From the perspective of sustaining high growth rates, problems arise. The amount nitrogen available for tree uptake remains high, but the amount of either Ca, Mg and K or all decrease. The plant requires relatively fixed ratios of nitrogen to individual base cations in order to make needles, bark, branches, stems and roots, and the uptake elasticity is not very large. Trees have prehistorically evolved to be adapted to a situation where nitrogen is always very limited, and where the best competitiveness is gained by taking it up as efficiently as possible. There are no prehistoric situations where the tree had to cope with more nitrogen than it could use for growth. Under the best case assumptions, and if growth could be optimized with respect to base cation availability, maximum harvest would be able to stabilize at 85 million m^3yr^{-1} . But unless acid deposition is reduced at least to the critical load, this would also imply that

the base saturation would reduce to very low values, the soil solution would in many places have high concentrations of Al that potentially make problems for tree roots and the stability of the ecosystem would be low. The base saturation is important as a source of short term resources to cover short term high needs of base cations. Such "loans" can be repaid in other seasons with weathering as usual but less growth. The historic data on leaching are derived from analysis of a few historical soil samples and historical analysis of lake waters. If it is assumed that the base cation budget is forced to balance by the year 2030 because of empty base cation reservoirs in the soil, then the whole tree harvest will increase yield for a short period while base cations can still be extracted for the exchange reservoir, but eventually growth will reduce to what is long term sustainable. Base cation leaching will be significantly reduced, if the acidity deposition could be reduced to the critical load for acidity. Then negative differences between supply and removal will be limited to approximately 32% of the area with present growth. Under such conditions 30% of the area could also sustain a significant increase of the growth. The present site yield index (Swedish; Bonitet) does not coincide with the amount of base cations available from weathering. The present site yield index is influenced by nitrogen availability, which for a large part of the area is not long term limiting. An alternative site yield index need to be developed, based on base cations, and the minimum of the two must be searched for. The calculation results shows that as the base saturation becomes depleted, base cation leaching will decrease, as has been observed for the Skåne Province in Southern Sweden (Barkman and Sverdrup 1996). Between 1983 and 1993 the base saturation decreased in the Province of Skåne, and the soil survey connected to the Forest Inventory in 1993 (not yet published, still under evaluation by the Agricultural University in Uppsala) also shows a decreasing trend in base saturation. The decrease in negative differences between supply and removal is a result of the lower availability of base cations as the reservoirs become empty. Not surprisingly, decreasing contents of Mg and K has been detected in the needles (Barkman and Sverdrup 1996). At present whole tree harvest is already practiced in a large part of Southern Sweden. This is at present done without return of base cation nutrients. This implies that these landowners are quickly consuming the nutrient resources of the soil and probably significantly reducing their fertility. This will also have repercussions on future forest property value (See the chapter on sustainable economy earlier). If the best case for uptake assumed cannot be realized, because the plant roots do not penetrate the soil 100% perfectly, because the annual weathering is not available for uptake, or because water limits uptake, then the amount that can be harvested will be proportionally less. On the average, weathering accounts for 60% of the base cation supply (The median/mean value of the nutrient base cation weathering rate in Sweden is $0.18/0.35 \text{ keq ha}^{-1}\text{yr}^{-1}$) and deposition account for 40% (median/mean value of the nutrient base cation deposition is $0.16/0.25 \text{ keq ha}^{-1}\text{yr}^{-1}$) of the supply. If the base cation deposition should decrease from an estimated $0.16 \text{ keq ha}^{-1}\text{yr}^{-1}$ at present to the estimated background of $0.12 \text{ keq ha}^{-1}\text{yr}^{-1}$, then that implies lowering the naturally sustainable harvest from 80-85 million m^3yr^{-1} to 70-80 million m^3yr^{-1} (Tab. 15.1 and Fig. ??). The present calculation assumes that Al has no negative effect on uptake efficiency, ($f(\text{BC}/\text{Al})=1$). This is a very optimistic assumption, and any negative effect of Al or pH on growth would tend to make leaching larger and uptake less, speeding up soil depletion. In these calculations we have assumed that the pool of organic matter in the forest soil was at steady state. In the long term this is almost correct, but in the short term it is most probably not. If the soil change from a net accumulation mode to a net loss mode for organic matter, large amounts of base cations can be made available. This effect should be able to offset the effect of base cation deficiency for several decades. It must be remembered that this is then another finite reservoir that also become long term depleted. We are not able at present to determine a priori the change in the rate of net accumulation or net loss from these pools with any certainty. This makes it very difficult to determine exactly when nutrient disturbances or effects of soil acidity will affect forest growth, except in very broad terms. A study of the results suggest that it may be possible to improve the present situation if high harvest volumes could be made to better agree with the sites with high weathering rate. Today there is a certain correlation between high growth and

| Nutrient | Present state | Best case possible % of area with larger removal than supply | 80% uptake efficiency | Whole tree harvesting |
|----------|---------------|---|-----------------------|-----------------------|
| Ca | 95 | 21 | 48 | 97 |
| Mg | 93 | 2 | 22 | 94 |
| K | 67 | 7 | 10 | 87 |
| BC | 97 | 35 | 83 | 98 |

Table 15.1: The table show percent of the forested area in Sweden with less supply than removal for base cations. The table show that the present situation with a high uptake and a leaching increased by acidification is not sustainable in at least 97% of the area. "Best case possible" is based on the assumption that trees will take up all nutrients above $5\mu\text{g l}^{-1}$ for Ca and Mg and $0\mu\text{g l}^{-1}$ for K, and that the roots have 100% efficiency in uptake in all percolate water. The "80% uptake efficiency" assume that the tree will have the same concentration limits as the best case, but that the roots only come in contact with 80% of the soil water.

high weathering, but harvest is not optimized with respect to weathering and sustainability. Under conditions with continuing acid deposition, such an optimization would be desirable.

15.5 Conclusions

There is no reason to think that a severe base cation shortage will be without effect on long term productive growth. On the contrary, *Liebig's law* is an absolute law derived from the universal law's of mass conservation and the basic principles of thermodynamics. It applies to forests equally much as it applies to all other biological growth. Swedish forest growth is not sustainable at its present rate of growth with respect to base cations. The reasons for this can be identified as two major causes;

- Increased depletion of the base saturation by acid deposition
- Increased uptake to growth removed with harvest in modern times
- partial misfit between high soil weathering rates and high harvest rates

Without high depletion rates caused by acid deposition, it would possible to sustain a steady state harvest of approximately 80-85 million m^3/yr . For higher rates of stem biomass harvest, addition of base cations will be necessary. The situation can be significantly improved by getting a better fit between high growth rate and high base cation weathering rate than today. It is important to realize that modern forest management is capable of increasing growth far beyond naturally sustainable growth for harvest. We can compare with an automobile; the maximum speed of the machine is not the same as the best traveling speed on a road. If whole tree harvest is practiced, then the demand for base cations and phosphorus per m^3 harvested wooden timber increase two- to three-fold, and approximately 40 million m^3 of forest timber from whole tree harvest can be sustained on natural resources alone. Thus there is no way the present harvest volume of 75 million m^3yr^{-1} can be sustained under whole tree harvest management, without addition of base cation nutrients. The calculations how in general terms that the nutrient mix

| Nutrient | Present state | Best case possible | Whole tree harvesting |
|----------|--|--------------------|-----------------------|
| | Average difference between supply and removal in $\text{kg ha}^{-1}\text{yr}^{-1}$ | | |
| Ca | -3.4 | 4.6 | -7.0 |
| Mg | -1.6 | 1.5 | -2.6 |
| K | -0.8 | 1.2 | -2.0 |

Table 15.2: *The table show the total average of the difference between weathering plus deposition, minus uptake and leaching. Larger removal than supply is expressed as a negative number. The values show that the present situation is not sustainable with the present conditions, that under the best case the weathering resources will be sufficient, and that the situation is grossly unsustainable with whole tree harvest without base cation return. Present state imply that the base saturation decrease on the average by $0.33 \text{ keq ha}^{-1}\text{yr}^{-1}$ and whole tree harvest $0.62 \text{ keq ha}^{-1}\text{yr}^{-1}$.*

added to the forest should contain primarily Ca, Mg and P, in the southern part of the country also K. We may conclude:

- At present, the geographical arrangement of harvest results in Swedish forestry being not sustainable with respect to nutrients
- Stem removal forestry up to 85 mill m^3 annually may be made sustainable if the harvest is carefully matched with the carrying capacity, dominated by the weathering rate
- At present, the practice of whole tree removal in southern Sweden is greatly unsustainable with respect to nutrients, and can be shown to lead to rapid exhaustion of the soils
- In Sweden, whole tree harvest cannot be sustainable unless substantial replacement of nutrients take place. Most urgent is K and P in the south, Mg and P further north.

The calculations stress the importance that sustainable forest management must include the management of nutrient fluxes and reservoirs. Forest growth for harvest must not necessarily be limited by natural supply of nutrients. Active management of nutrients for growth is a great economic possibility, if it is carefully performed within the environmental constraints and optimized with respect to the properties and boundary conditions of the locality. It may seem futile for the Swedish forest owner to in the short term to try to compete with farming of fast growing eucalyptus and similar trees in tropical climates, but once the insight to what sustainability is has been won, it will cease to be a long term threat. Such short term successes as the fast growing eucalyptus farming will soon find itself caught up with the fundamentals of sustainability. Much of such farming occur on highly weathered lateritic or quartzitic soils and red earths with very low intrinsic weathering rate. At present the forest farmers are irreversibly cashing out their soil nutrients, leaving a depleted and empty soil behind. Only when located on high weathering, low silicic volcanic soils, such forests may survive for any longer period of time. This insight will allow us not to indulge in such short term opportunism, nor to fear it, but stay long term sustainable in our own strategy.

Chapter 16

General conclusions

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16.1 Statements and conclusions

We may conclude that the data collected at Asa, Jämjö and Stenbrohult provide unique possibilities for actually assessing forest sustainability from more than one aspect. The SUFOR-programme has produced a set of new models that may become future sustainability instruments, after due adaption and additions to make them more user-friendly. The design for such tools are already taking shape. The integration process has lead to clash of ideas, vigorous discussions and a rethinking of established paradigms. The cooperation between Lund University, Lund Institute of Technology, Swedish Agricultural University, Swedish Environmental Research Institute (IVL) and Gothenburg University has been necessary for this development process. In the synthesis of the work, we have identified probable conflicts between goals if every aspect of the future forest management ought to be sustainable. These conflicts have to be further investigated to find ways to minimize or resolve these conflicts. The first conflict type depend on if the forests are looked upon as static or a dynamic ecosystems by the planners and researchers. As a biological system the forests are always dynamic and the development depend on changes in the natural as well as anthropogenic impact factors. This is especially true in a long-term perspective, i.e. over several hundred of years, Fig. 16.1. For example, the static demand from a sustainable forest biodiversity perspective to keep large areas with unaffected natural forests are hard to preserve within a dynamic approach. The second types of conflicts highlight the problem where the goal to keep one biological process sustainable contradict other sustainability goals. Such conflicts are: An increased area with recurrent burning activity to enhance forest biodiversity vs the long-term economizing of nitrogen or the goal to maintain the soil as a carbon sink. The third conflict depend on the definition of good economy of sustainable forest management. The traditional definition of good economy is to evaluate the cost/benefit balance of different management activities from planting to clear cut. In a long-term sustainable perspective the cost/benefit balance have to include the costs of a depletion of soil nutrients.

We may make some important conclusions after the first phase of the SUFOR Programme:

1. Sustainable yield

- (a) The standard yield is NOT equal to the sustainable yield, but rather a systematic overestimate. The standard yield represents the maximum growth traditional management can produce. The sustainable yield ins the long term acceptable amount that can be harvested without causing adverse effects.
- (b) Sustainable stemwood yield as m^3 stemwood ha^{-1} could be determined at Asa based on nutrient availability and tree cover properties. A principal method for estimating

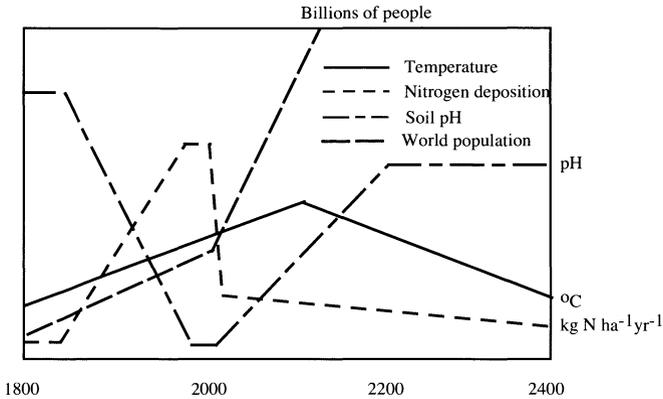


Figure 16.1: *Face the facts ! We have experienced very large changes in the basic conditions (Climate, soil chemical conditions, world population, nitrogen availability) for sustainable forestry, and we must expect equal or even larger changes in the next 300 years to come . This will present us with a number of difficult problems, and facing those will become the existential questions for future sustainable forest management, and it will necessitate that the strategies adopted must be strategies that can adapt to such changes.*

the sustainable yield has been developed. Scaled down to the level of a small property of 100 hectares, a sustainability assessment would cost less than 3,000 Swedish kronor, plus a day or two of work time for the forest owner. The assessment would be valid and useful for several decades to come.

- (c) The most critical factors for nutrient sustainability are: Weathering in the collective rooting zone of the stand. This is depending in order of importance on:
- rooting depth (depend on soil depth, groundwater level and tree species)
 - soil content of minerals,
 - soil texture,
 - soil moisture,
 - soil stoniness,
 - soil temperature.
- (d) Leaching from the soil is dependent apart from the soil conditions mentioned above, on the net acidity input.
- (e) Base cation deposition depending mainly on location in the country and vegetation type.
- (f) Nutrient removal at harvest depend on:
- standing stock and tree age,
 - nutrient content of removed material,
 - harvesting strategy (stem only/whole tree/time of the year).
- (g) Nutrient sustainability cannot be robustly tested, but the components of it can. The long term biogeochemical consequence of it's exceedance can be observed and used for testing.
- (h) The traditional forest yield does not reflect the sustainable yield at a certain stand in a satisfactory way. Integration of trees with deeper roots mixed into the stand, significantly increase the sustainable yield of the stand. The project could determine

Structural view of the prediction/management system within SUFOR

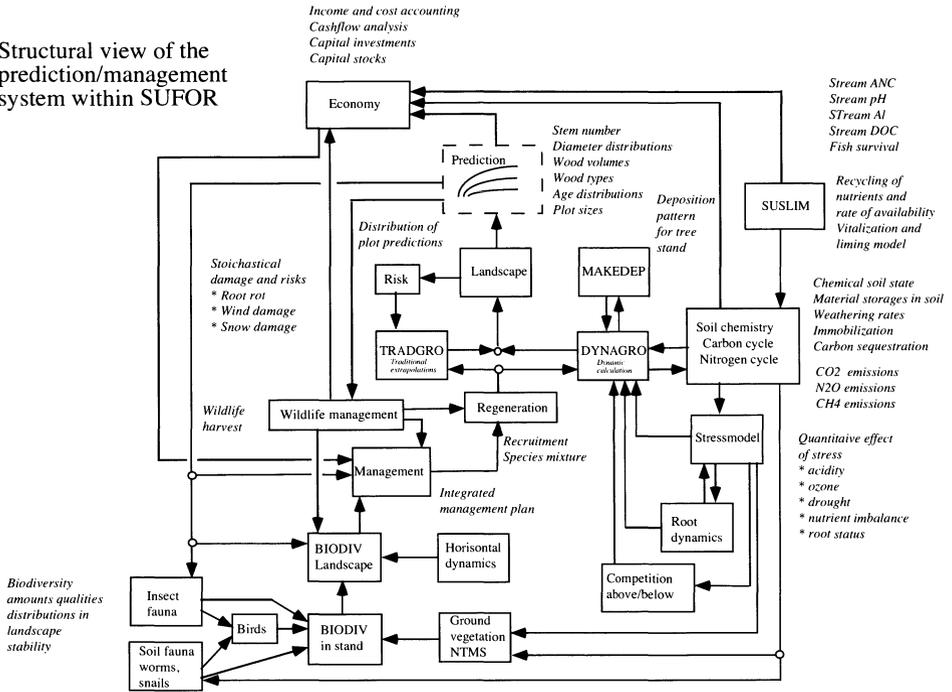


Figure 16.2: It is important to construct maps over the problems investigated. In SUFOR, mental maps were used extensively to map the problems and to help simplification in order to obtain simple operational models.

that trees with deeper root systems for nutrient uptake have a higher nutrient sustainability potential than more shallow-rooted trees. Norway spruce may be brought to grow faster than deciduous trees, but this is irrelevant for the sustainable growth which is determined by other parameters.

- (i) Stem harvest must under certain circumstances be assisted with modest limestone and non-nitrogen nutrient additions to restore nutrient pools. Present base saturation levels in Asa are low, and some assistance to natural recovery would be desirable.
- (j) Whole tree harvest without nutrient replacement is always very harmful and will deplete the soil badly in short time. The depletions risked this way may be serious enough to lower the yield capacity of the soil significantly, also ruining land property value. Whole tree harvest remove much nutrient in low quality biomass and within the sustainability limitation, this represents a less efficient harvest strategy.

2. Sustainable forest vitality

- (a) The change in basic conditions is a threat to forest vitality. The fundamental processes for forest production and biodiversity are currently changing owing to the long-term impact of excess nitrogen, the ongoing soil acidification and a raised tropospheric ozone level as well as an expected future climate change. In response to these threats there is a real risk for decrease in wood production and quality as a

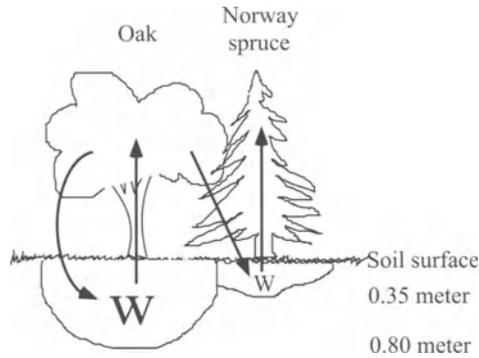


Figure 16.3: *THINK SYSTEMS*; The deeper roots of the deciduous tree take up large amounts of base cations to supply the larger nutrient circulation in the canopy. According to our hypothesis, in a mixed stand in the upper soil layers caused by the higher rate of nutrient circulation. At Asa as well as at Jämjö, this hypothesis could be shown to be acceptable. The deciduous tree help the conifer tree by giving access root help. The consequence of our hypothesis is that effective fine root depth become one of the most important sustainability parameters. Management of average fine root depth in stand and on property become a main tool for sustainable nutrient management when the margins are small. Overcrossing of roots in adjacent tree's root systems become an important parameter. This is more important in Europe where the trees do not use *allelopathy* as a *competitive weapon*.

consequence of changed ecological conditions likely to effect both growth and susceptibility to pest and pathogens or weather extremes

- At present, the soils in southern Sweden accumulate large amounts of nitrogen. From 1850 to 2000 the soil store of nitrogen has more than doubled, from 2 ton/ha to more than 5 ton/ha. Soil acidity contribute to the nutrient imbalance in the soil as well as to changes in the quality of the runoff water. Declining forest floor C/N ratios indicate increasing risks for nitrate leaching. All forest soils south of a line from Kalmar to Göteborg have the potential to leach substantial amounts of N to the water ecosystems.
 - Although the precursors of ozone may have decreased in Europe by the year 2010, the effects the ozone levels are unclear. NO_x reduction may lead to increases in ozone in some situations. Furthermore, rapid increases in Asia will increase the ozone levels in the northern hemisphere, which in turn may affect the ozone levels in Europe.
 - Predictions suggest that southwestern Sweden will experience a warmer (+0.5-1°C) and wetter climate (+15%) during the next 150 years, while the southeastern Sweden will become warmer (+1-3°C) and dryer (-10%), thus threatening tree species on the border of their climate range, such as spruce.
 - The Swedish forests can sequester large amounts of C in the short term; 8 million tons of C year⁻¹ in standing trees, and approximately 2.5 million ton C year⁻¹ in the soil litter. This is however not permanent. Management strategies that sequester carbon in the forests more efficiently than present must be developed.
- (b) The change in tree vitality depend on two fundamental processes which are affected by the change in environmental conditions

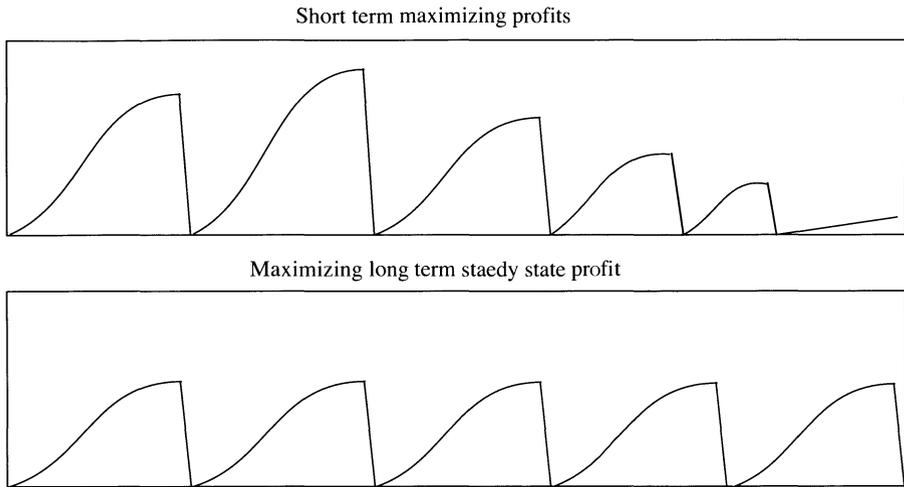


Figure 16.4: *THINK LONGER*; A much longer time perspective in forestry is necessary. Over-exploitation of the site will over time erode the production potential. Instead of maximizing the growth rate for the moment or even a single rotation, it is necessary to integrate the maximized production over a long time, many rotations. The production must in the long term accumulate to the sustainability potential. This applies to nutrient resources as well as other resources such as biodiversity.

- Nutrient imbalance

Imbalance of macro and micro nutrients in the tree is related to increased frost sensitivity and predisposes the tree to heavier infections by pests and pathogens. There seems to be a difference among species which of the nutrients is the most affected by environmental changes and this has to be further investigated.

- Changes in carbon balance

Current ozone levels as well as nutrient imbalance change the carbon balance of the tree through decreased root/shoot ratio, production of defense substances and winter hardening. The changes in the growth pattern affect the uptake of nutrients and water which, by a negative feedback mechanism, further reduce tree vitality.

(c) The possibility of countermeasures. Too counteract negative effects on forest vitality, two ways are possible:

- A negative nutrient budget for a specific stand can be countermeasured by an input of deficient nutrients. Forest liming and different kinds of vitality fertilizers have been tested. The site specific nutrient imbalance should be considered in the choice of countermeasure method. If vitality fertilisation will be used continuously in a large scale, the environmental costs for society, like the effects of mining and transports must be included in the assessment of sustainable forestry.
- A change from monocultures to mixed species stand is a way to "use" forest ecology to improve the nutrient balance for the whole stand by utilizing the increased rooting depth. This approach has the added benefit of being a long-term method to counteract nutrient imbalances.

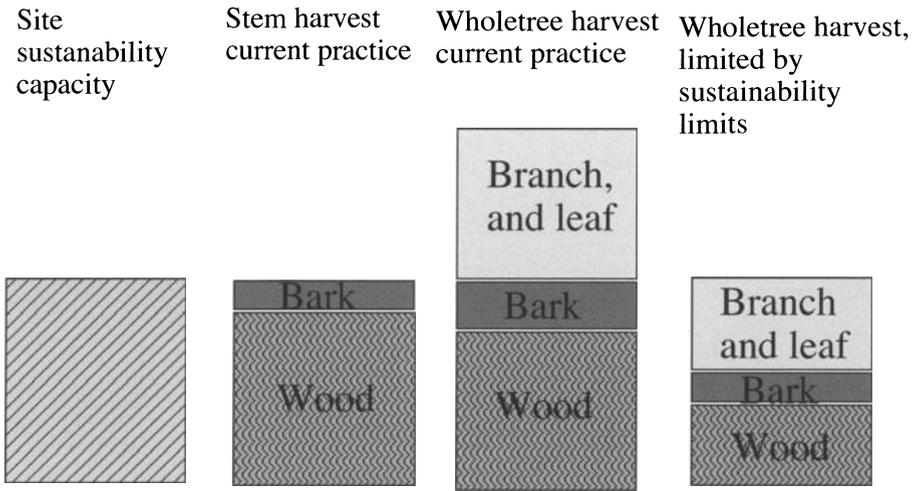


Figure 16.5: *THINK SUSTAINABILITY LIMITS*; For production within the sustainability potential, whole tree harvest is a poor economic strategy. Removal of slash, which is a low value product, will require much more nutrients per unit weight. The fraction wood, the high value product, makes the largest fraction of the harvest under a stem harvest strategy. Present whole tree harvest, almost everywhere it is practiced, grossly overexploits the soil resources, unless the excess amount is added.

- Monitoring of forest condition. Monitoring of various aspects of forest vitality and system sustainability is vital in order to assess the combined effects of e.g. air pollution, future climate change and the outcome of forest policy and actual *de facto* practices within forestry. Long data series are also important from a research points of view, in particular when the lifespan of a tree generation is considered. Current integrated national monitoring programmes operate with to few monitoring sites where comprehensive data sets, covering both abiotic and biotic parameters, are collected to be fully useful. Similarly, the lack of recurrent regional forest decline surveys with high spatial resolution is unfortunate.

3. Sustainable biodiversity

- (a) Biodiversity can be adequately protected by protecting and preserving it in number of core areas in the landscape. These areas must have a size above a minimal critical area of at least 10 hectares and make up at least 7% of the total landscape area.
- (b) It is possible to protect biodiversity by creating refuges in areas with small or no sustainable productive value. Thus detection of soils with low sustainability potential for stemwood production can become important stocks of essential biodiversity in the landscape. A number of indicators of sustainable biodiversity has been proposed.
- (c) The predicted future vegetation changes caused by climatic and pollution stress changes will imply that deciduous trees will regenerate easier in the forests and make up a larger part of naturally regenerated forests. A warmer climate calls for a paradigm change in forestry and a challenge to sustainably increase productivity and profitability within the forest of mixed species.

4. Social sustainability

- (a) Management can well adapt to profitable harvest and management that at the same time is seen as aesthetically pleasant by the public. Well managed forests are seen as aesthetically pleasant by the public.
- (b) A mixed forest with differentiated age structure is seen by the public as being more desirable than monocultures and monotonous single age/single species stands
- (c) Biodiversity is socially sustainable when it at the same time preserved aesthetic values in the landscape. The public observes and appreciates a biodiversity of vegetation and faunal elements, whereas diversity in insects are often seen as a surplus of "bugs" and "pests". Functional and structural properties are not observed. The aesthetic components of biodiversity must be built such that the unperceived but important elements are preserved.
- (d) The societal value of recreation in the forest accounts for 30% of the value, whereas traditional forestry accounts for 65% and hunting approximately 5%.

5. Economic sustainability

- (a) Sustainable forestry based on mixed stands can be optimized to an economic profitability equal or larger than the economic profit of a sustainable forestry based on Norway spruce monoculture.
- (b) Present economic profit are at present based on unsustainable growth in 70% of all stands and requires mining of nutrient resources that cannot be replaced at the rate of present removal.
- (c) Sustainable forestry in Southern Sweden is possible within the present profitability requirements.
- (d) The "Norway spruce monoculture paradigm" is not sustainable over large areas in Sweden. A change is necessary for present profitability to survive into the next century.
- (e) Conditions for sustainable forest profitability is larger in southern Sweden than in the north.
- (f) We need to adapt and further develop planning and prediction instruments.
- (g) Economic conditions:
 - If a nutrient accounting management is adopted, then any production may be made sustainable with respect to nutrients. Vitalization and whole tree harvest may even be used as a tool to eliminate nitrogen from a forest ecosystem.
 - Traditional interest calculus is not a valid investment analysis method over one or several forest rotation. The argument that traditional economic do not know any other method is not a valid argument for using invalid methods. Alternative calculation methods are available and have been applied in the SUFOR programme.
 - Forest management needs to modify its accounting principles. Hidden incomes and hidden costs needs to be included in the income accounting. It is important to include the management costs for soil resources, as well as costs involved in environmental and social obligations.
 - Hidden assets and hidden liabilities needs to be included in the balance estimates. It is important to increase or decrease in internal stock of nutrient resources, as well as internal movements between stocks. Economic liabilities associated with long term responsibility obligations must be internalized and included in the balances.
 - The adoption of sustainable productive yield must be introduced in management planning, and the active management of any difference between this and the

operative yield must be properly included in the income/cost statements as well as the changes in stocks taken to the balance.

- An analysis of the future market suggests that there will be an increased demand for wood products over the next 3 rotations (300 years). In the short term, half a rotation, both printing paper and sawed products will show a positive development in consumption. An ongoing consolidation in the industry may in the very short term (10 years) force raw wood prices down. During 1985 to 2000, the hardwood sawmill products and white copying paper were the fastest growing sector, paralleled by diminishing standing stock in many places. In the long term, paper is expected to be used for a wider range of products, and consumption is expected to rise with population and GDP at least until 2150.
 - The constraint of sustainability remove the advantage earlier enjoyed by Norway spruce monocultures, putting deciduous tree stands and mixed forest stand on a equal or better position in terms of value generation.
- (h) Operational tools for the forest owner
- A simple tool for nutrient sustainability estimation is available and will be provided free of charge to any forest owner.
 - It is possible to establish the sustainable production potential at a forest property for a cost of approximately 5,000 SEK per 100 ha property, the input data gained will be valid for the next 500 years.
 - The sampling activities at Asa has resulted in knowledge of the necessary sampling density in order to be able to interpolate the results on maps, the uncertainty of the parameter values have been determined and the necessary information for making statistically correct and significant upscaling was obtained. We have determined that the critical sampling distance between points at Asa is approximately 700 meter, or one sample per 25 hectare land. Thus for a 100 hectare property, about 2-4 soil samples will suffice to establish the sustainability potential with good accuracy.

16.2 Risk management

1. In a forest management for biodiversity sustainability, the density of older living and dead trees as well as the frequency of disturbances regimes will increase. There is a risk for an threat to tree vitality owing to an increased population of pest and pathogens and facilitated dispersal.
2. The private forest owners are the key actors in the work towards a future sustainable forest management in southern Sweden. To stimulate this change, a larger assorted range of wood products for the market is important as the forest/farm unit then will produce a higher profit. Specific risks can be identified on the property level:
 - There is a risk that the current biodiversity policy formulated by the Swedish Government, NGO's and international certifiers gives a top-down perspective on the work for higher biodiversity, with associated risks for poor acceptance and owner participation. The forest owners, private as well as others, must be involved in the process to maintain or increase the forest biodiversity.
 - With an increased demand for more fire disturbances there is a risk for raised costs for the private forest owner. The owner has to keep watch over the burnt areas both in space and time. With few persons available, there is always a risk for the fire to spread to stands outside the planned burning area. Other economic considerations which ought to be further investigated are fire intensity vs increase of root rot and

decrease in organic concentration in the soil as well as the size of the burnt areas vs storm damages.

We can conclude that the forest management procedure must be an iterative, adaptive method of designing the management plan. It is important that the proposed plan is iterated several times round the planning cycle and repeatedly tested against the criteria, in order to optimize the approach towards full sustainability and simultaneously maximum profitability. When conflict of criteria or conflict of interests occur, a structured model for handling these must be employed, setting clear priorities (Fig. 4.1).

In the project, we were able to move from a scale of single stands up to landscape level at 3,200 ha. In that range, the approaches have full interpolability and thus we may down-interpolate from landscape scale into any property or stand. Additional sampling on the property will increase certainty and quality of the sustainability assessment. The gap to the national level at 22,700,000 ha could not be continuously bridged in the first phase, but data came forth that will definitely make this possible. With the presently available databases (2001) this is now fully feasible as well as the degree of uncertainty is known and controllable.

16.3 The future

We have identified several gaps that can be the focus for future research:

1. Develop knowledge about soil root occupancy and average rooting depth for different species, stand types and soil properties as well as knowledge how these conditions affects the uptake of nutrients and water.
2. Develop biodiversity models and assessments into an integrated biodiversity assessment including fauna, vegetation, scale and structure. Being able to cope with climate change will be the single largest challenge to all biodiversity management research, regardless of other factors involved. The prospect of climate change may overturn all existing approaches for protection and conservation of biodiversity and makes the research for new approaches urgently needed.
3. Develop knowledge about the effects of climate change and nutrient availability on the carbon allocation of the trees including models for pollutant fluxes to mature trees and stands. Present global climate models have, when at all, insufficient feedbacks to the nitrogen cycle and sustainable forest management, and present scenarios needs complete redoing.
4. Test sustainability assessments in the field.
5. Develop system knowledge about effects of countermeasures to preserve tree vitality during the whole life cycle of the tree.
6. Develop economic models for small private properties and non-industrial management.
7. Develop integrated forest property models (100-10,000 hectares) for all production aspects.

This will be elaborate in the application for the next phase of SUFOR.

Chapter 17

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