Towards the Sustainable Use of Europe's Forests – Forest Ecosystem and Landscape Research: Scientific Challenges and Opportunities

Folke Andersson, Yves Birot and Risto Päivinen (eds.)

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IUFRO International Union of Forest Research Organizations

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Introduction

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Networking European research on forest ecosystems is highly desirable

Scientific research in Europe is organised on the basis of national institutions such as universities and research organisations. Over the last 20 years, the policy initiated by the European Union through 5 consecutive Framework-programmes for RTD has resulted in the building of scientific communities in various disciplines, structured in networks between teams across Europe. Before being transformed into an ambitious political concept by the Commissioner Philippe Busquin, the European Research Area (ERA) was already becoming a reality. To achieve ERA, a new Framework-programme for RTD, as well as other new instruments, have been launched by the European Commission – the 6th Framework Programme.

Sustainable Forest Management (SFM) has become the pillar of forest policy at national, European and global scale, in the context of post UNCED Conference (Rio), and World and European processes (MCPFE). However, SFM is a complex issue. A sound implementation requires improved knowledge within a multidisciplinary framework. Research can contribute, to a large extent, in providing the rationale and the background for such an integrated management of forests, making the forest sector a key component of sustainable development.

The situation of forest related research, particularly in the field of sustainable forest management, has been assessed in the context of a COST study on "the inventory of forest research capacity in Europe (1999)". This work has shown that despite significant research forces in Europe on biophysical and socio-economic disciplines, fragmentation and dispersion of research teams and laboratories are real handicaps. There is obviously a need for concentration of means, in particular in scientific disciplines highly demanding in skills and equipments: forest ecosystem research is certainly among those. The launching of the 6th FP together with its new instruments, paves the way to a better coordination in this field. Among the new instruments of FP6, in particular the major ones; it appears that the option "Network of Excellence" (NoE) is better suited to the purpose of assembling in a durable manner scientific competences across Europe for advancing knowledge, rather than the option "Integrated Project" aimed at producing clear and quantified deliverables through integration across technical areas, activities (research, technology, dissemination and various bodies).

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A network of Excellence on forest ecosystems, the Centre for European Forest Science: background, objectives and organisation

The history and the genesis of the preparation of a NoE in the field of forest ecosystem and SFM can be summarized by recalling a few milestones. In 1991, the Ministerial Conference on the Protection of Forests in Europe passed a resolution (among others) approved by 25 countries, stating that coordination of forest ecosystems research efforts should be strengthened within and between countries. One concrete outcome in the implementation of this commitment was a concerted action under FP4 called EFERN (European Forest Ecosystem Research Network). Building on this background, the COST Action E25 called ENFORS (European Network for long-term Forest Ecosystems and Landscape Research) was initiated in 2001. ENFORS was a perfect platform for preparing the 6th FP and an expression of interest was submitted to the EC by ECOFOR/ENFORS in the course of 2002.

After the summer 2002, the first draft of the work-programme in priority area 1.1.6.3 "Global change and ecosystems" was issued. It was noticeable that topics, such as forest ecosystems or sustainable forest management were left out of the first call. EC representatives confirmed this at the open seminar "Forest research and the 6th FP – Challenges and opportunities" held in Paris on 25/11/2002 and co-organised by ECOFOR and EFI in the context of the IMACFORD project. A side meeting held the next day concluded positively on the relevance of preparing a NoE on Forest Ecosystems research, ECOFOR and EFI acting as coordinators. A timetable was set up defining the main tasks and milestones.

Following contacts with a broad range of institutions throughout Europe, a meeting was held in Paris on 25 February, 2003 between representatives of Institutions. An agreement was reached on the main principles of the project as well as on its overall focus: forest management aspects, multidisciplinarity, landscape, approach, and society aspects (goods and services). A preparatory structure (coordination, drafting group, steering committee) was set up and tasks (lobbying, enlargement to other countries) were allocated. Further on, a second meeting took place in Vienna on 27 April, 2003, which allowed reaching an agreement on the pre-approval of the draft programme for a NoE, and on the composition of an advisory group. The discussion also dealt with the preparation of the Tours Symposium and the selection procedure and criteria for participation.

This preparatory phase has led to principles and steps as follows:

- A NoE called "Centre for European Forest Science (CEFS) aimed at: i) reaching a Pan European network of lead scientists, ii) paving the way for multidisciplinarity, iii) benefiting from well resource and highly skilled labs
- CEFS could accommodate a force of about 500 researchers within 40 institutions
- A draft programme has been set up and circulated
- An inquiry on research capacities to be involved in the project has been carried out

The **main scientific challenges** of the tentative NoE are as follows:

- Generalisation and integration of approaches leading to a functional understanding
- Making multidisciplinarity a reality by involving transdisciplinary teams in project planning, implementation and dissemination
- Improvement of the conditions for multiscale approaches

The preparatory work has also allowed identifying the **main scientific objectives**:

- three background research areas on: i) ecosystem research, ii) environmental economics, iii) forest management and practices

- research oriented to problem solving (policy, planning, techniques): i) multifunctionality of forests, ii) land-use and role of forests in the landscape, iii) long-term trends in the environment
- innovative tools and methodologies in terms of: i) experimental resources, ii) observation, measurement and monitoring, iii) scientific tools

The **Joint Programme of Activities (JPA)** is focused on the integration of activities related to the scientific challenged. This will be achieved through mutual programming in the three research areas, developing common resources and exchange programmes, creating common labs, networking existing facilities and merging forces. JPA also will include jointly executed research aimed at developing key projects, with structuring effects, targeted to problem solving in relation to SFM. Last but non least objective of the JPA is the spreading of excellence among members and non members of the consortium.

It is intended to achieve **excellence in research** through the share of scientific knowledge and resources between skilled teams, the common experience of applied problems, a broad bio-geographical representation and competences allowing to take into account the specificities of various forest ecosystems throughout Europe.

In terms of **management of CEFS**, it is intended to handle the management of the three research areas at the institution level. The problem-oriented research will be classically implemented through strongly coordinated projects, while common management of scientific tools will be looked for. Of course governing bodies of CEFS will be put in place such as: overall coordination, steering committee, advisory group.

The Tours symposium – a major stage in the process of building the Centre for European Forest Science

The objectives of this Symposium in Tours were mainly to provide a scientific forum where the latest results in forest ecosystems research can be presented, as well as to discuss the need for future collaboration in order to support decision-making and practical measures. Another main issue is to discuss and support the preparation of the NoE whose applied goal is sustainable forest management. The symposium is structured in 2 parts: scientific fora and a session devoted to the preparation pf the NoE.

The scientific fora are focused on the 3 research areas: i) ecosystems functioning including biogeochemical cycling and functional ecology, ii) environmental economics and sociology addressing multi-functionality (economic and social values) and public policies (market organisation, participatory policies), iii) forest management and practices (design of forest policies, answering management questions, improvement of techniques and practices. The fora are planned with 2 plenary sessions on forest managements and practices, and environmental economics and sociology, while they are 2 parallel sessions on biodiversity and ecosystem functioning. These sessions have to identify potential topics for the NoE.

The preparation of the NoE is addressed through several sessions: a plenary aimed at proposing a strategy on how to proceed, another plenary on the methodological aspects, and 3 parallel workshops aimed at identifying the topics for the NoE.

Although uncertainties are still remaining about the timing of occurrence of a work-programme in FP6, which will address forest ecosystem research and SFM issue, it is believed that the preparation of CEFS should be pursued in the coming months on the basis of the principles elaborated in Tours.

Forest Management and Practices

Forest Planning in Europe: State of the Art, International Debate and Emerging Tools

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Abstract

Forest planning is the discipline through which forest policy is expressed and management choices are made. It has long been based on a simple system, formalised in the late 18th century, in which the main aim was the production of wood. Forest planning must now extend to other objectives and to other – often more generic – objects, in which the forest plays a role together with other shareholders. European stakes are examined in the light of the international debate, however, without going into detailed national particularities. Emerging methods, concepts and tools are also presented.

Keywords: forest planning; sustainable forest management; multifunctionality; multidisciplinarity; tertiary society

1. Introduction

1.1 Background

People have been adjusting their surroundings according their own needs since the Neolithic times as expressed by Lamy (2001): "The only species capable of creating its own environment by transforming the one placed at its disposal". The intensity and nature of human intervention in their environment, all interfering with natural cycles, have thus been adjusted in accordance with the expected usages sometimes appearing contradictory (Farrell et al. 2000).

The forest has also followed this pattern, as illustrated by Léonard (2000) as he emphasises "the strong influence of societies' living conditions on the future of surrounding wooded areas". Forest planning is an example of this (Farcy and Devillez 2003). The meaning of the term "development" (in French *aménagement*, which is closer to *design*] has evolved and is

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understood in many ways: limited or extended; indicative or normative; strategic or operational, even regulatory. Hence, the English expression "forest planning" infers more of a notion of groundwork than the generic French term of *aménagement*.

In the late 18th century, France and Germany started to develop forest planning as a discipline to meet the needs for the increased control over the forest as a resource. Forest planning found itself rooted in two sets of principles: the principles of cultural order in the etymological sense of wood culture and the principles of economic order linked to this production (Hartig 1805; Huffel 1926). The purpose of the traditional forest planning methods is thus to organize sustainable production and yield in time and space. Associated functions such as soil protection or the preservation of hunting areas were ensured implicitly by the maintenance of a wooded area.

Two centuries later, as the post-industrial society developed, tertiary activities and consumption started to play an increasingly important role in the European economy, especially in the most developed countries. This evolution had so many social, political and cultural consequences that the term 'Tertiary Society' is often used (Fourastié 1969; Petit 1988). The environment's role evolved considerably. Nature, and the forest in particular, were not exceptions to this trend, which sparked many international debates in which the forest played an important, or even preponderant role.

As the only species capable of being conscious of its acts, humans have been bound to shoulder their responsibilities in modifying the surroundings. With the Rio Conference, the early 1990s marked the start of an official internationalisation of debates on the matter. Countries officially accepted the importance of the roles and uses of the forest, which had thus far been considered as secondary (Brédif and Boudinot 2001; Mårell et al. 2003). Soil protection and water quality, the preservation of biodiversity and the quality of landscapes, the stocking of carbon, not to mention the recreational and cultural values of the forest, found themselves the subject of increased attention in international texts. In addition, the process of forest retrocession in the countries moving towards a market economy has added yet another specific element to this picture.

2. New stakes

As a crossroads for the expression and development of contradictions between stakeholders with more or less explicit rights, a place of arbitration between the objectives and a crucible of interdisciplinary, forest planning remains an essential discipline but it is forced to evolve and adapt (Andersson et al. 2000; Laroussinie and Bergonzini 1999).

Although not completely ineffective, the modelling of forest planning based on a simple system centred on wood production has clearly become insufficient. This can be argued e.g. with the questioning of the wake effect, according to which careful wood management is also beneficial to environmental protection, employment and public access (Peyron 2002).

The shortcomings are to be found essentially in the fact that by tackling new goals and objects, forest planning, which must endeavour to find an equilibrium between interests that are often conflicting, comes up against other temporalities and other natural and social loci. These characteristics, observed throughout European forests, mean that the issue can be ranked among those phenomena that come under the heading of complexity (Barraqué 1997).

The riches of the contrasts between the Mediterranean and the boreal region, not to mention the mountain ranges and urban forests, boast, however, certain specific cases in which there is a unique twist to the changes, which in turn require specific management solutions (Buttoud 2000; Garcia Lopez et al. 1999).

2.1 Other spatio-temporal scales

Traditional forest planning develops in a space circumscribed by the boundaries of the wooded area and of the property in the meaning of the civil code. The spatial element that constitutes its elementary unit is the forest stand, defined by the trees that make it up. When, as now, other objectives are sought, forest planning must take into account the new ranges of spatio-temporal scales in which the natural or human phenomena in question develop (Sverdrup and Stjernquist 2002). This can lead to the simultaneous presence of different spatio-temporal models.

This challenges the integrating status of the forest that monopolises wood production; indeed, for other functions such as the ecological or social functions that develop according to a spatial continuum, the forest is integrated into more generic concepts such as the landscape or the watershed, in which it interacts as a partner.

2.2 New human components

The tensions between the forest as “shared heritage of humanity” and the “appropriate forest space” where the owner reigns supreme illustrate the rupture and the diversification of the owner’s prerogatives to the benefit of new communities of users (Brédif and Boudinot 2001; Comby 1997). The crisis is particularly acute in terms of the legal and organizational zoning of territories, the net limitations and spatial partitioning of which are poorly adapted to suit the complex flow chart of rights, usages and encumbrances (Farcy and Devillez 2003).

Legislatively, organisationally and politically speaking, contradictions are emerging, causing conflicts over areas of competency, for example between naturalists and urban planners.

2.3 New types of decisions

Having been organised in accordance with culturally and economically-based principles, the decisions that are part of the traditional forest planning procedure have found, within rationalist approaches, appropriate tools and methods that are still being developed (Gong et al. 2001; Kangas and Kangas 2002; Pukkala 1998; von Gadow 2001). These are based on the stable result of deductive reasoning, the heir of modes of thought born of economic theories such as the concept of optimisation.

However, the current decision-making context is less linear than such an approach might imply. It is often characterised by co-decision and conflict, the multiplication of decision-making levels, structural uncertainty caused by a lack of understanding of the processes, a narrowing of the decision-making horizon and the ensuing need for permanent adjustment, as well as the presence of non-market values and qualitative criteria. This decision-making context, which is poorly adapted to a strictly rationalist approach, seems better suited to incremental-type approaches (Buttoud 2000). This line of reasoning opens up a vast field of research, which begs to be pursued.

3. Current trends in Europe

In the face of these new stakes, European foresters have progressively introduced modifications and adaptations to their practices. The trends currently at work in Europe are all part of the efforts to implement sustainable development and in particular the three pillars of Sustainable Forest Management: multifunctionality, dialogue and follow-up. Integrated in the concept of multifunctional management, they are the European counterparts of American Ecosystem Management (Brussard et al. 1998). In practice, these concepts are translated and developed in three different ways which will be described below.

3.1 Spatio-temporal and organisational hierarchy

The proof of "a relative decline in classic planning at the property level or at least in the ambitions harboured at this scale", the general trend observed in Europe is towards the "development of directing diagrams to the detriment of forest management plans that could end up being reduced to a timetable of felling and other work" (Subotsch-Lamande and Chauvin 2002). Indeed, there is now an increasing movement towards rebalancing the levels of planning and creating hierarchies by the definition of more marked intermediary levels between the national level and the property level: the regional level, the clump level and the landscape level (Bachmann et al. 1999; von Gadow 1995; Sverdrup and Stjernquist 2002).

"The regional level makes it possible to transpose national objectives into technical directives… The small clump level or the landscape level is that of land use planning wherein maps of objectives and functions can be traced and can constitute the basis for local directing diagrams… The landscape level thus becomes the scale of preference for integrating ecological and cultural values" (Subotsch-Lamande and Chauvin 2002). The concept of habitat in the sense set out in European Directive 92/43/EEC can also be integrated as a possible management unit.

3.2 Participative approach

The participative approach tradition is already well established in certain European regions, while public participation is now considered essential by all forestry authorities whether this be via information, education or concrete involvement in the decision-making (Bettelini et al. 2000; Solberg and Miina 1997; Subotsch-Lamande and Chauvin 2002).

However, power is a central concept in participative forestry; the representativeness of the players, the transparency of the processes and the availability of resources to facilitate the process are often underestimated (Buchy and Hoverman 2000). This element is all the more important to take into consideration given that the issue at stake is little known and its effects poorly controlled.

3.3 Follow-up

The current climate of uncertainty, whether subjective or objective, has resulted in a narrowing of the decision horizon and of decisions' duration of validity; climate change could for example affect the distribution, phenology or adaptation of species (Levêque 2001). In planning, this requires the integration of permanent evaluation and readjustment procedures. Having been formalised in company management in the form of a Balanced Scorecard (Kaplan and Norton 1998), the use of methods relying on criteria and indicators has become common in forestry as a means to meet the need for standardisation or as a part of international certification procedures.

The concept of progressive management has also gained ground. Conceptualised in America under the label of Adaptive Management, it aims for interaction between research and management thanks to permanent follow-up, with a view to adapting management on a continuing basis to meet the demands of society and the risks of nature (Lessard 1998). While well adapted to large spaces, this concept becomes more difficult to implement when the spaces in question are reduced and fragmented and when, as a result, recourse to the compensation process becomes less effective. Moreover, there is still no solution to the problem of diversity in response times, which can sometimes be in the distant future (Moir and Block 2001).

3.4 Comments

As suggested by the presentation of current trends, these are still imperfect, sectorial, and they appear to be the fruit of modular thinking that still needs to be integrated. Whereas forestry is undergoing profound change, serious challenges subsist.

That of multidisciplinarity and the necessary connection between knowledge acquired from different scientific fields; the idea is to update and complete the principles of classic forest planning whereas, for certain new functions, the forest is no longer the only entity at stake.

This being said, the need for multidisciplinarity is far from unique to forestry. The resurgence of human ecology (Leroy 2001) or of the concept of anthroposystem (Levêque 2001) or again the hesitations of landscape ecology between geography and biology (Bastian 2001) are all signs of the need to connect the science of nature and the science of societies and to bring the interfaces under control. For its part, the environment’s economy finds itself faced with the contradictions of abiding by the same reasoning for both market and non-market, which illustrates the difficulty of connecting areas in which the players have completely different value scales (Bontems and Rotillon 1998; Clays-Mekdade et al. 1999). Finally, this need for multidisciplinarity is clearly illustrated by the emergence in America of the concept of biocomplexity, which “arises from the multitude of behavioural, biological, social, chemical, and physical interactions that affect, sustain, or are modified by living organisms, including humans. From cells to cities to the global ecosystem, all systems associated with life exhibit biocomplexity” (Covich 2000).

Another challenge is that of continually attempting to reconcile, in a climate of incertitude, the growing need for a better knowledge of natural and social processes, and the need to provide satisfactory solutions for the people in the field who have to make decisions and manage on a daily basis. Therefore, while it is true that the spatio-temporal and organisational hierarchy of forest planning was necessary operationally speaking, it would also be useful to reconnect space and time in a more functional perspective with a view to providing appropriate management and action solutions. New spatial functionalities such as the transition zone, the neighbourhood and proximity should thus be developed. From the temporal point of view, the natural dynamics of vegetation, for a long time interrupted or disturbed by the forester, offers a new, much less linear horizon. Current uncertainty, on the other hand, forces a narrowing of the planning horizon and recourse to the concept of outlook (Gallopin et al. 1997).

Last, and we would deem this the major challenge; the forester must juggle between a resource-oriented approach – with which he is sufficiently familiar – and a player-related approach. Forest planning, which is in essence at the frontier between the natural and human sciences, sees the latter occupying an increasingly important place.

4. Emerging tools

4.1 Functional approaches

Far from being emerging since it dates back to the 1950s (von Bertalanffy 1956), the systemic approach is of particular relevance today given the stakes forestry must confront. Ollagnon (1984) had already used it in the 1980s within the framework of a patrimonial approach to quality management of natural milieus. Sverdrup and Stjernquist (2002) use it as the basis for the development of principles and models for sustainable forestry in Sweden.

It seems sound today, within the framework of a functional approach. The identification of new objects on which the planning bears and the understanding of the *functional systems* of which they are part is a crucial step that could indeed benefit from a systemic approach: a methodology based on the organisation of the level of complexity with a view to increased efficiency of action (Laszlo 1981). At the basis of this approach lies the acceptance of this complexity and of the ensuing incertitude, as an underlying hypothesis. "To analyse and understand a complex object in terms of a system, you must start by defining its scope, i.e. by drawing a virtual border between this object and the rest of the world and considering it as a whole, made up of hierarchical and interdependent sub-systems" (Mazoyer and Roudart 1997). The spatio-temporal and social framework thus defined makes it possible to identify tools and methods in line with the level of complexity (Arnould 2002; Buttoud 2002; Farcy and Devillez 2003).

The functional dimension of this approach is illustrated by the concept of landscape, which can be seen in a unique light once it is defined on the basis of the observed system: "the landscape criterion means that one is considering relative spatial relationship on a plane, or a loose equivalent of a plane, as the organizing principle for proximity of parts of the observed system" (Allen 1998). This means that the landscape thus defined does not include a priori a concept of spatial scale, because it is linked to a level of organisation that cannot be a priori linked to a particular spatio-temporal scale. The ecology of landscape fits in with this functional definition (Farina 1998; Forman 1995).

This type of systemic approach is efficient for building models at a specific level of organisation. This being said, the formal coupling between levels of organisation and the integration of the spatial position of objects must be examined in greater depth. For example, spatio-temporal conceptual models can offer data modelling facilities for the description and management of complex objects with spatial and/or temporal components (Parent et al. 2000). Modelling and simulation by *multi-agent* approaches and *cellular automata* are starting to give results, the former for simultaneous modelling of social and natural dynamics, the latter for the dynamics of vegetable systems. As the heirs of the formalism of artificial intelligence, they make it possible to represent sophisticated behaviours and to simulate their consequences (Schmidt-Lainé and Pavé 2002).

In the context of forest planning, the use of this type of modelling and simulation can be used, on the one hand, to understand complex environments and especially to acquire a better *understanding* of the processes, and, on the other hand, to insert them into collective decision-making processes in complex situations. At this stage, they can be used to facilitate *dialogue* between players and as a negotiating tool (Ferrand 1999; Etienne 2003).

4.2 Data, information and knowledge

From whatever angle one looks at planning, there is no denying that it is a major producer and consumer of information (Laroussinie and Bergonzini 1999). Taking the three pillars of

Sustainable Forest Management into account only reinforces this point-of-view. In particular, the constraints created by the need for follow-up and dialogue increase the importance of the process of generating knowledge, in which data and information intervene and which consists of progressive research for meaning.

Upstream of this process, new technologies have proven increasingly efficient in *collecting* descriptive data. The use of images produced by very high spatial resolution satellites or of airborne laser scanning technology make it possible, for example, to optimise data acquisition, which was long limited by the sensors' resolution (Alonso et al. 2001; Hoss 1997; Kristof et al. 2002; Walter 1998). However, integration with traditional field collection techniques is still essential (Dees et al. 2001).

The benefits of producing and *processing information* via Geographic Information Systems (GIS) have already been well established within the framework of client-server paradigms. These systems have proven useful in the analysis phase, as well as during spatial stratification in relation with environmental issues, not to mention as a visual support, sometimes three-dimensional, for negotiation assistance (Bettinger and Sessions 2003; Subotsch-Lamande and Chauvin 2002). The emerging applications and functions of Web Mapping could prove useful in the area of information sharing between institutions or partners.

Indeed, further down the line, integration into an Information System defined as "an organised combination of people, hardware, software, communications networks and data resources that collects, transforms and disseminates information in an organisation" (O'Brien 1996) becomes essential. Nevertheless, the extension and generalisation of Information Systems require taking into account the constraints of both the technical and semantic interoperability of the systems (Richards and Reynolds 1999).

Associated with the development of information technologies and the avalanche of data that are produced, but also with the current context of incertitude that encourages one to reduce risk and incertitude by increased and improved knowledge, the discipline of *Knowledge Management* is emerging in environment applications (Tochterman 2003). It is defined as a "collection of methodologies and strategies, finally based on Information Technology, to encourage the flow of knowledge between people in an organisation" (Kolp 2002). This characteristic concept of the tertiary societies in which we evolve illustrates "the current trend in problem solving; long based on pragmatism and the intellectual tradition of analytic rationality, it relies on a better integration of the human factor, and particularly its psychological, social and cultural contributions. That means a better appreciation of man's ability to use the information available to make sense and to mobilise knowledge" (Nonaka and Nishigushi 2001).

American foresters are attempting to integrate it into the concepts of ecosystem management and adaptive management, which should benefit from an increased understanding of the process of knowledge generation (Riemenschneider and Potts 2003).

5. Conclusions

To be able to evaluate forest planning in Europe under the undergoing profound changes of the world, it was necessary to discuss many other things than the forest itself, which helped to get a clearer global picture and to define the issue more clearly.

The challenge forest planners face today is a part of the broader issue of the cohabitation of the functions of the economic sector of primary production with the functions of a tertiary society: the forest has a monopoly in the former case, and is a part of a whole in the latter case.

By accurately defining the forest from the different points-of-view involved, all of which are legitimate, and by carefully controlling the different interfaces, it should be possible to pave the way for multidisciplinary and thus to identify the missions and levels of competency of each party. The social and natural diversity of European forest constitutes a trump in this conceptual approach.

As the link through which forest policies are expressed and through which management choices are made, forest planning could find a part of solution in this approach to the major changes it is facing, as well as a specific role relying on a mastery of the forest ecosystem in the broad sense of the term.

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Modelling at the Interface between Scientific Knowledge and Management Issues

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Abstract

Forest management deals with ecosystems developing over large temporal and spatial scales. Models enable us to link ecological information with silvicultural knowledge and to translate research findings into practical implications, hence allowing the forest manager to analyse the long-term consequences of silvicultural regimes in a highly dynamic and permanently changing environment.

Most forest models developed so far are stand level approaches. However, issues like environmental impacts, biodiversity, wood markets, and stakeholder involvement, require approaches at multi-level temporal and spatial scales. Also different modelling approaches exist. The more empirical, the more accurate the models are in their short-term predictions, but empirical models are highly depending on the data-set used for parameter fitting and are therefore limited with respect to application in other growing or environmental conditions. Process models, in turn, do include functional relationships with e.g. the environmental conditions, but generally lack a direct applicability for forest management. As a consequence, the many process-based models developed pas decade were almost exclusively used for ecological and ecophysiological research, whereas even the newly developed management tools still largely consist of descriptive relationships.

The term "hybrid models" was introduced more recently to refer to approaches that contain both causal and empirical elements at the same hierarchical level. The strength of this modelling approach lies in combining a mechanistic approach with allometric relations. The hybrid approach (also referred to as scenario studies or scenario modelling) seems to offer the best opportunities to become a forest management support tool.

One of the key issues in forest management in Europe today is how to account for changes in environmental conditions determining forest growth and development, while at the same time coping with the changing demands from society for forest goods and services. There is an increasing need of considering the role of forests on larger time and space scales. Issues

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like biodiversity and water resources require that both the forest and landscape ecologist and the forest (landscape) manager can estimate the role of changes in environmental conditions (societal, physical) as well as predict the impact of management operations on the ecosystem and landscape level. Scenario modelling is a promising tool here as well. By including a spatial scaling approach, e.g. using land-use cover types of forest inventory data, these models can support the evaluation of long-term consequences of different types of land-use, management regimes, and land-use or forest policies.

The importance of models as support tool for answering current and future questions by forest policy makers and forest managers will further increase. Models are unique tools to clarify concepts, to determine consequences of choices, and, finally, to support decision making. In all circumstances, to develop models suitable for modern days forest stand, enterprise, and ecosystem management, close co-operation between scientists (the model developers) and forest managers (the model users) is required.

Keywords: modelling; scenarios; forest management; hybrid models; up-scaling

1. The changing role of forests and forest management

Over the past centuries, forests produced wood and forestry was largely about growth and yield (Olsthoorn et al. 1999; Pretzsch 2001). Over the past decades, matters have grown complex. In addition to providing wood as a raw material for society, other forest functions became increasingly recognised as being important, and traditional concepts of sustainability in forestry were expanded from sustained yield to management for sustainable multi-functional use. This was partly invoked by over-exploitation, degeneration, and disappearance of forests world-wide, partly by the increasing importance of providing a range of goods and services other than timber, and partly by the increasing awareness that forests play key roles at local, regional and global scales in regulating water, nutrient and carbon flows. It became clear that forests affect atmospheric composition (pollution, C-balance), water resources, biodiversity, global climate, gene flows, etc., and that forest management also affects developments on spatial scales beyond the stand level.

With the shifting emphasis in forest management from sustained yield (wood production) to sustainable natural resource management or ecosystem management (Nabuurs et al. 1998; Monserud 2003), the emphasis on use of natural processes in forest management increased, in order to minimise costs, and to warrant those aspects of forest use that are associated with naturalness and biodiversity (Olsthoorn et al. 1999; Nabuurs et al. 2001). Many definitions of sustainability and sustainable forest management were developed. For the purpose here, it is useful to realise that for those aspects of sustainability that can be made explicit and tangible, simulation models can be used to analyse the consequences of the sustainability concept in relation to forest management.

As a result of the developments mentioned, sustainable forest management nowadays tries to incorporate a wide range of goods and services, and often is no longer restricted to the stand or forest enterprise level only: forest management needs to consider processes and conditions at the landscape level and beyond, e.g. in case of biodiversity or climate change issues.

Forest management deals with ecosystems developing over large temporal and spatial scales. Forest ecological and silvicultural research provide fundamental knowledge on these ecosystem dynamics. Models enable us to link the ecological information with the silvicultural knowledge and to translate research findings into practical implications. Models hence allow the forest manager to analyse the long-term consequences of silvicultural regimes in a highly dynamic and permanently changing environment, not in a prescriptive but in an analytical and exploratory way (‘what-if’-questions).

2. The historical development and application of forest growth models

Models of forest growth, from hand-drawn diagrams to sophisticated computer models, have been and still are important forest management tools as they provide guidelines for decision making. Many models were developed during the past decades (see Figure 1). Four major developments affected forest growth modelling the past centuries: 1) the silvicultural focus moving from even-aged monospecific stands towards mixed-species stands, 2) the growing interest in incorporating causal relationships in models, 3) the changing goals of forest management, and 4) the increasing availability of computers. The history of forest growth modelling can hence not simply be characterised by a continuous development of improved models. Instead, different model types with diverse objectives and concepts have been developed over the past decades simultaneously (Pretzsch 2001; Porté and Bartelink 2001). Most forest modelling occurred and occurs at the stand scale. Two main applications can be distinguished: growth and yield estimations including the prediction of the effects of forest management practices, and studies of forest dynamics and succession including the effects of natural disturbances and stand structure (Porté and Bartelink 2001).

The model types developed vary from highly empirical to more mechanistic approaches. In empirical models, growth estimations are derived from time-dependent statistical relationships between tree or stand characteristics and external variables or forcing functions. Many empirical models have been applied in forest management (Pretzsch 2001). Process-

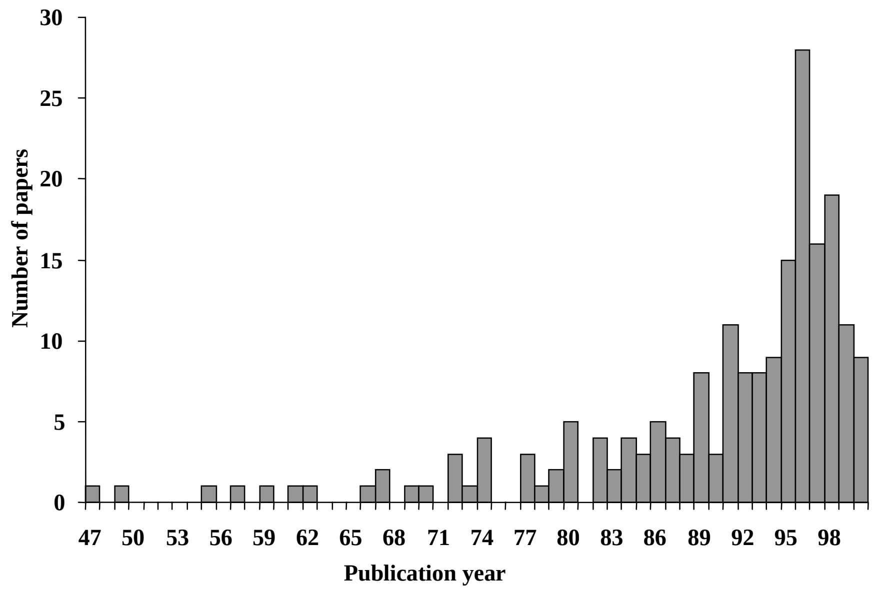

Figure 1. Number of publications on forest growth modelling at the stand level, over the past decades (from Porté and Bartelink 2001).

based or mechanistic models, on the other end of the model spectrum, simulate the behaviour of an ecosystem by identifying the functional components and their causal interactions, as well as the causal relationships with the environment. These models are primarily scientific tools, providing formalised statements of hypotheses and a framework that gathers different pieces of information and knowledge. Mechanistic and empirical models have long been considered separate tools for separate aims. Mohren and Burkhart (1994) concluded that many gaps between biologically-based process models and management-oriented growth and yield models existed. Differences in focus were and are largely related to differences in application. The use of models as tools in research aims essentially at contributing to the understanding of how a system works, not at prediction of systems behaviour. Typically, such models aim to include as much causality, or assumptions on causality as possible, sometimes at the expense of accuracy in parameter estimation. Models aimed at prediction consist more often of summaries of large amounts of data that have been collected in the real world. Descriptive models are then used to depict these data in summarised form, so they can be used as a reference for decision making in conditions comparable to those in which the underlying data were collected. In both cases, models and systems analysis can be used as tools to bridge the gap between science and practice (Rabbinge 1986).

Most forest models developed so far are stand level approaches, aiming at supporting decision making about the protection, treatment, and utilisation of forest resources at the level of forest enterprises (Porté and Bartelink 2001; Pretzsch 2001). Measurement series from permanent sample plots provide an important data source for this. However, issues like environmental impacts, biodiversity, wood markets, and stakeholder involvement, require approaches at multi-level temporal and spatial scales (Figure 2). To facilitate forest decision-making both at the stand level and at the landscape level, models are needed that are capable of integrating different spatial and temporal scales and that include ecological and socio-economical aspects as well.

3. Application of models in forest management decision making

3.1 Stand level modelling

Models that are closest to practical applicability are those that produce information directly linked to forestry practice, like data on growth and yield, wood quality, or sensitivity against wind, fire, pest, and diseases (Pretzsch et al. 2002), and that do take environmental conditions into account. Both mechanistic and empirical approaches are used to support forest management decision making (Korzukhin et al. 1996; Peng 2000).

The more empirical, the more accurate the models are in their (generally short-term) predictions, provided the predictions are within the data-domain that the model was derived from. However, empirical models like traditional growth and yield models are highly depending on the data-set used for parameter fitting and are therefore limited with respect to application in other growing or environmental conditions (Mohren and Burkhart 1994; Monserud 2003).

An important side-effect of the increasing belief amongst scientist as well as forest managers that empirical models no longer suit forest management decision making, is that it directly threatens the maintenance and accessibility of the large data sets that exist in Europe as a result of long-term permanent growth and yield research. These data form the back-bone of empirical models like yield tables. Considering that the models are no longer relevant might also result in ignoring the accompanying data sets, which would be a great loss; these

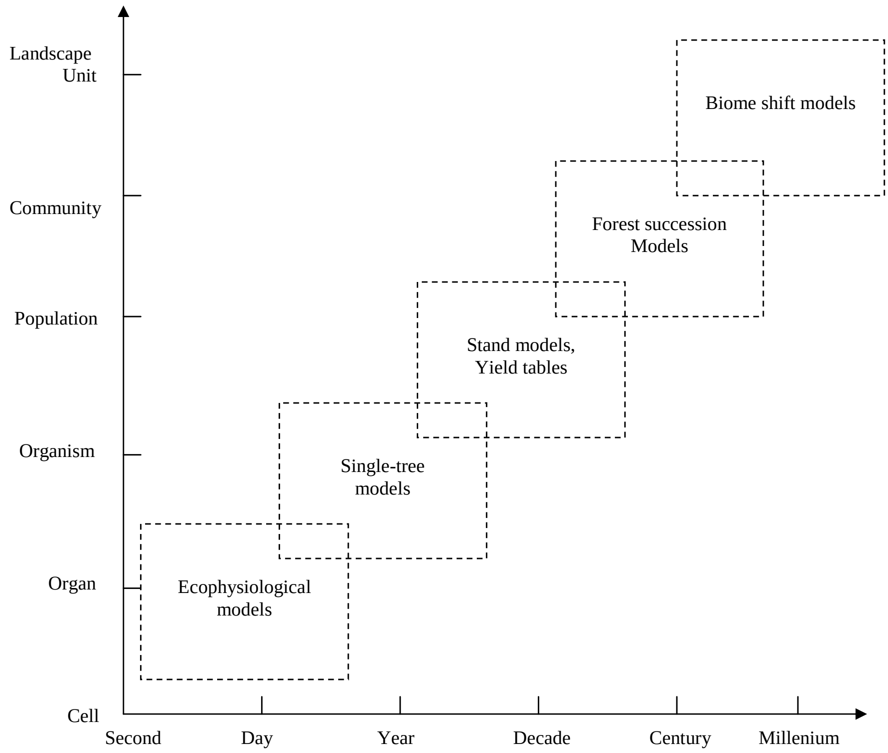

Figure 2. Spatial and temporal dimensions of processes in forest ecosystems, and models with increasing aggragation, from ecophysiological models to biome shift models (Pretzsch 2001).

data contain valuable information, like historical growth trends that e.g. can be used to study the impact of past changes in environmental conditions (see Spiecker et al. 1996). Moreover, the historical data can also be valuable for validation of mechanistic, process based models!

Growing conditions are gradually changing; events like the increased N-deposition in western-Europe, the rising CO_2-levels, and climate change, will strongly affect composition, growth, and development of forests (Spiecker et al. 1996). This points at the need for mechanistic approaches. A drawback of such process-based or mechanistic models, however, is that they “over-do the job”: for many questions related to forest management problems, a high level of ecophysiological detail is not necessary (and not feasible). Process-based models are generally considered to contain too many variables, and too complex variables, to be useful as such for forest management. Besides, many parameters are difficult to catch, and many have highly uncertain values. Pure process-based models are hence rarely used as practical tools for forest management (Mohren and Burkhart 1994) and there is a strong belief among forest managers that the conventional empirical approach to growth and yield predictions is superior (Mäkelä et al. 2000).

Forest managers to date hence have to cope with an almost paradoxical situation. On the one hand, there is a increasing uncertainty in the long-term (e.g. regarding environmental issues), pointing at the need to include more causality in the forest models. On the other hand, this also implies more uncertainty about the validity of the results. As a consequence, forest

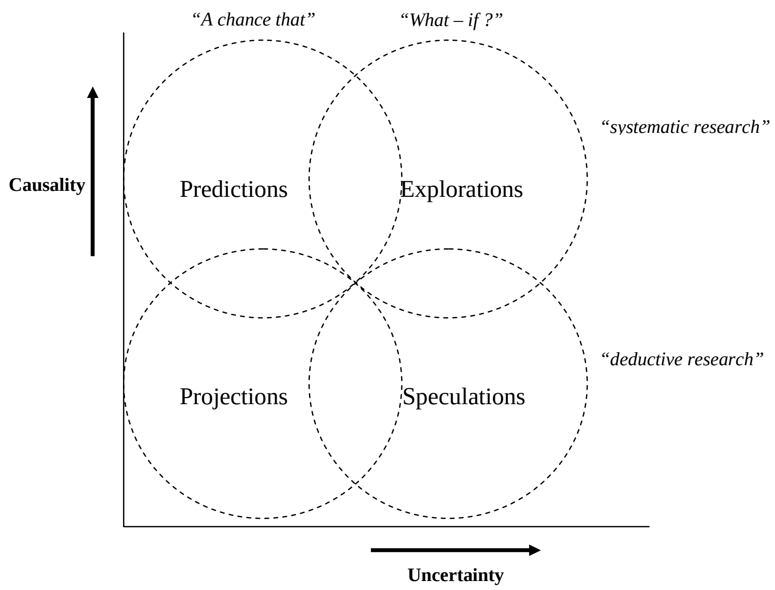

Figure 3. Typology of research aimed at predictions of the future. If uncertainty in the system and model is apparent, "what-if" type questions can be addressed. If uncertainties are small the probability of future events can be assessed. If causality of the model is prominent, more systematic future research is possible. If causality is lacking, only regressive or deductive methods are available, leading to projections or speculations of future events and conditions (Becker and Dewulf 1989, cited in: Van Ittersum et al. 1998).

managers might choose to rely more heavily on empirical data sets derived from long-term growth and yield study plots, hence implicitly disregarding the potential longer-term effects on forest development and the consequences for forest management. This is illustrated in Figure 3, which shows four different ways of dealing with future questions; predictions of future developments can only be made when uncertainties are small and a causal model is available. If causality is lacking, however, descriptive or deductive methods are possible, seducing forest managers to extrapolate from past growth data.

In the late 1990s, researchers started to realise the potential mutual benefit of sharing and exchanging model philosophies, i.e. blending empirical and mechanistic approaches (Mohren and Burkhart 1994; Korzukhin et al. 1996). Bringing in more causality by adding mechanistic relationships to an empirical model would improve the model validity and adaptability, because then the reliability of the model depends on the state of knowledge of physiological processes and responses to the growing conditions of the species involved rather than on a statistical fit to a particular set of data (Mäkelä et al. 2000; Robinson and Monserud 2003). The term "hybrid models" was introduced to refer to approaches that contain both causal and empirical elements at the same hierarchical level. The strength of this modelling approach lies in combining a mechanistic approach with allometric relations, using long-term data series for model evaluation. Actually, this is not so much a new concept rather than a description of the evolution of forest modelling. Korzukhin et al. already stated in 1996 that neither pure process-based models nor pure empirical models exist, but, instead, that all models can be placed on a imaginary axis running form 100% empirical to 100% mechanistic.

Though the desire to have more process information in models for forest management has lead to the development of more generalised and simplified process models (Johnsen et al. 2001), to date still a gap exists between the modelling approaches; the many process-based models developed pas decade were almost exclusively used for ecological and ecophysiological research, whereas the newly developed management tools still largely consist of descriptive relationships. Mechanistic models applied for management decision making should contain as little process-detail as possible, while maintaining a minimum level of causality. Mäkelä et al. (2000) states that the practical implementation of process-based models and causal thinking in forest management issues would be accelerated if it becomes generally accepted that empirical models can be improved through the incorporation of causal relationships, and that mechanistic models can be improved through the inclusion of empirical elements.

The hybrid approach seems hence to offer to best opportunities to become a forest management support tool. In current studies, the term scenario studies or scenario modelling is used as well to indicate this modelling approach. Mohren (2003) describes scenario models as a combination of process-based models of forest growth with management strategies (e.g. in the form of forcing functions). In this way, the mechanistic model is used to scale up results from plot level studies to stand and forest enterprise levels (Figure 4). A few process-based models in a scenario-setting are close to or already reached the operational level in supporting forest management (e.g. 3-PG from Landsberg et al. 2001; EFISCEN from Pussinen et al. 2001).

Choosing the scenario modelling approach requires data from long-term research plots (Rastetter et al. 2003). Therefor, existing long-term plot data should be more intensively exploited and access to them needs to be improved. The set-up of meta-databases on growth and yield data (e.g. CIFOR) and tree allometry (European COST activity) are important conditions for this.

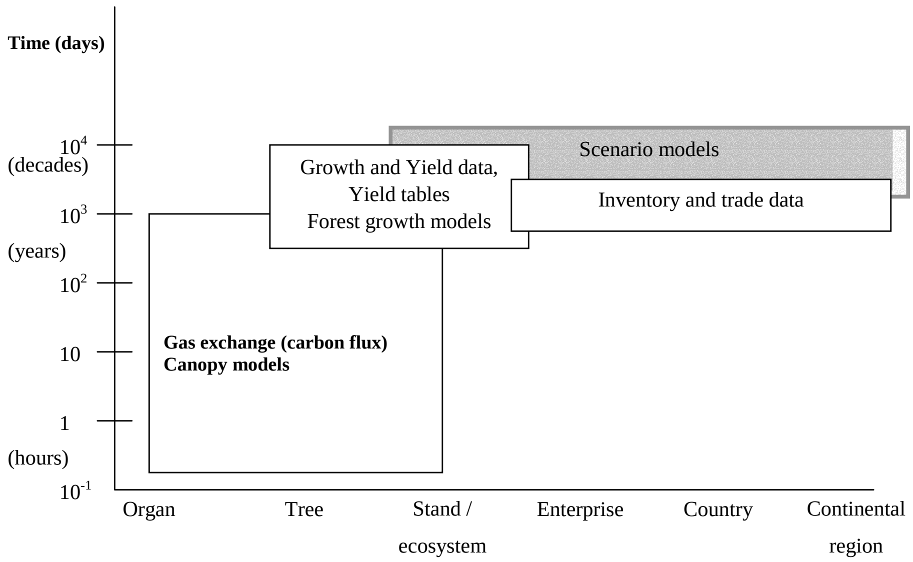

Figure 4. Resource assessment and modelling approaches at different temporal and spatial scales, and the role of scenario approaches (Mohren 2003).

3.2 Necessity for up-scaling

One of the key issues in forest management in Europe today is how to account for changes in environmental conditions determining the forest growth and development, while at the same time coping with the changing demands from society for forest goods and services. There is an increasing need of considering the role of forests on larger time and space scales. Issues like biodiversity and water resources (IPCC 2001) require that both the forest and landscape ecologist and the forest (landscape) manager can estimate the role of changes in environmental conditions (societal, physical) as well as predict the impact of management operations on the ecosystem and landscape level (Nabuurs et al. 1998). Most existing forest models operate at the forest enterprise level (i.e. from several hectares up to a few hundreds of hectares). Up-scaling, i.e. extrapolating outside the initial data range, would theoretically be possible provided the model would contain a large degree of causality. However, this would require an exhausting amount of input data, from which many are uncertain or even unknown. Using descriptive approaches to achieve up-scaling could solve that problem. The challenge is to find the optimal way between detailed mechanistic ecosystem modelling on the one hand, and between larger-scaled spatial information on land-use patterns and changes.

Scenario modelling is a promising tool here as well. By including a spatial scaling approach, e.g. using land-use cover types of forest inventory data, these models can support the evaluation of long-term consequences of different types of land-use, management regimes, and land-use or forest policies (Mohren 2003). The mechanistic models used in the scenario study should be summary models, i.e. reduced ecophysiological complexity while maintaining sufficient causality. Using GIS-tools would further increase model applicability, enabling one to achieve geographically explicit information. This is of particular interest for Europe, where current agricultural policies are under discussion, where timber markets are under pressure, and where forest management has to deal with conflicting interest, both locally and though different organisational scales. Meanwhile, there is an strongly increasing interest of stake-holders and the general public in forest, forestry, and landscape, demanding that forest management decisions are well justified (e.g. making long-term consequences into account), explained and communicated. Such challenges need larger-scale scenario analysis that enable the linking of existing knowledge and various approaches for answering the management and policy questions.

4. Concluding remarks

Scenario modelling, relying on a hybrid modelling approach, offers the best opportunities to develop forest models suitable for supporting management decision making. Hybrid approaches, nevertheless contain some pitfalls. For instance, the accuracy of process models can be improved when more functions and parameters are included, but increased complexity reduces the generality and broad utility of the model (Johnsen et al. 2001). Another remark is that hybrid approaches might loose clarity in model structure and error propagation. The causality in the model easily can be overwhelmed by the empirical elements: does it make sense to calculate photosynthesis rates in a model that also uses site index curves?

Many models exist nowadays. Near future research should hence focus on the accessibility, adaptability and exchangeability of existing models rather than on new models (Johnsen et al. 2001; Pretzsch et al. 2002; Robinson and Monserud 2003). Robinson and Monserud (2003) identified 10 criteria for determining the adaptability of forest models. They concluded that, from the models they examined, non was capable to meet all adaptability requirements. Johnsen et al. (2001) stated that incorporating causality increases transportability of a model, but that increased complexity reduces a broad utility. These results clearly indicate that there is much to gain when accessibility, adaptability and exchangeability of models becomes a greater concern of forest modellers. Initiatives like electronic publishing (FBMIS) and like putting models on the internet largely speeds up the exchange of knowledge and increases the opportunities of discussing and exchanging model concepts.

In conclusion, forest models form an interface between forest science and forest management applications:

1. Models make ecological knowledge more easily accessible for forest managers by summarising and quantifying ecological processes and tree and forest conditions.
2. Models enable the linking of ecological knowledge with silvicultural knowledge.
3. Models allow the forest manager to analyse the long-term consequences of silvicultural regimes using scenario analysis.
4. Modelling, in the search for data, includes data-mining, hence increasing the value of long-term permanent plot data series and stimulating their accessibility.
5. Models, both at the stand and at the landscape level, are useful to support decision-making in forest management, both for *prediction* (within the data range, using especially data-rich empirical models) and *exploration* (outside the data range, emphasis on mechanistic elements).
6. Exisiting long-term growth and yield data sets are extremely valuable to test various types of models, and provide a consistent historical bio-assay on forest structures and forest dynamics under the conditions at the time of sampling.

The importance of models as support tool for answering current and future questions by forest policy makers and forest managers will undoubtedly further increase. Models are unique tools to clarify concepts, to determine consequences of choices, and, finally, to support decision making. In all circumstances, to develop models suitable for modern days forest stand, enterprise, and ecosystem management, close co-operation between scientists (the model developers) and forest managers (the model users) is required.

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A Moving Target: Forest Growth in a Changing Environment – the Role of Long-Term Dynamics

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Abstract

European forests have changed because management induced drastic shifts in tree species and tree age composition, as well as due to changes in site conditions and areal extent of forests. Management affects site productivity by altering nutrient cycling and by changing competition for light, nutrients and water. The observed increase in wood production and growing stock at many forest sites has been accompanied by a loss of biodiversity, a shift to non-site-adapted tree species, and an increase in biotic and abiotic risks. Changing forest conditions and changing needs of society require a widened scope for forest management.

Knowledge on the effects of management options on functions and processes in forest ecosystems and their environmental services is rather limited. Processes in forest ecosystems are operating on multiple spatial and temporal scales. Response time of forest ecosystems to disturbances ranges from the short-term up to decades and even centuries, depending on the condition of the system and the type, intensity and duration of the external stimuli. The actual state of forest ecosystems largely depends on processes of the past. As an object of research, forest ecosystems are not easy to approach.

For the control and management of sustainability in the development of the forest resources, specific scientific knowledge needs to be considerably deepened and enlarged: the relevance of tree species composition, mixture and canopy structure for the impact of environmental changes, their dependency on site conditions and their modification through management need to be analyzed on a broad scale. Sensitive parameters for assessing the state of the forest ecosystems with respect to their resilience towards environmental threats need to be identified. Retrospective observational studies provide a means for describing short- as well as long-term forest dynamics and the complex structure of multi-scale relationships. Results of the studies on long-term dynamics and the associated key processes will be of significant value for improving our understanding and hence for improving the predictive power of process-based modelling approaches.

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1. Forests in the face of change

Forest ecosystems in Europe experienced drastic changes due to centuries of large-scale human activities associated with the use and sometimes overuse of natural resources. Intensive land-use changes have largely modified the face of European landscapes and forests (Williams 1990; Bork et al. 1998; Kirby and Watkins 1998; Küster 1998; Glatzel 1999; Hunter and Schuck 2002). The management of forests affects tree species composition, stand structure and hence site productivity by altering nutrient cycling and by modifying competition for light, nutrients and water. Forest ecosystem processes are furthermore altered through human induced changes in disturbance regimes (e.g. pest control, fire control) which together with the long-lasting preference of certain tree species have led to new types of forest ecosystems(e.g. Jahn et al. 1990). The increase in wood production and average growing stock which is observed at many forest sites in Europe (Kauppi et al. 1992; Kuusela 1994; Spiecker et al. 1996) has been accompanied on a large scale by a loss of biodiversity and a shift to non-site-adapted tree species both resulting in a decrease in stability against biotic and abiotic disturbances (Spiecker 2003).

Forests provide well-recognized goods, including timber, fuel-wood, and raw materials for industrial products. Beside their productive function forest ecosystems also provide services such as maintaining biodiversity, water and soil protection, mitigation of floods, cleansing of the atmosphere, sequestration of carbon, aesthetics and recreation (Arbeitskreis Zustandserfassung und Planung 1974), services which have only recently received increased recognition (Linckh et al. 1997; Weidenbach 2001; Gadow et al. 2002; Johnson et al. 2002).

The management and use of forest resources can directly alter the functioning of forest ecosystems and the services they provide. However, human activities also indirectly affect goods and services provided by forests through changes in the environmental systems that interactively control forest ecosystem functioning. Observed and anticipated directional changes in the atmospheric, climate, soil, and hydrologic systems due to e.g. air pollution, atmospheric matter deposition and CO_2 enrichment are creating novel conditions that affect the functioning of forest ecosystems and their stability (Kellomäki et al. 2000; Kirschbaum 2000; Watson et al. 2000; Dale et al. 2001; Karnosky et al. 2001; Hunter and Schuck 2002).

For managing and sustaining forest ecosystems under changing environmental conditions and in changing socio-economic contexts – with altered needs of society, increasing public awareness and number of stakeholders – new management approaches and a widened scope of forest management is needed. The challenge for sustainable forestry today is to identify and define the attributes of forest ecosystems that are ecologically and societally important and to optimize and sustain ecosystem goods and services in the face of change (Chapin et al. 2002; Mårell et al. 2003). To achieve these goals, a more comprehensive understanding is needed of:

- the role of forest properties for ecosystem functioning,
- the relation between ecosystem functioning and goods and services provided by forests,
- the interactions and trade-offs between goods and services provided by forests,
- the interrelationships between land-use history, management practices and ecosystem functioning.

Improved scientifically based knowledge is necessary in order to implement ecological principles into forest management planning, management strategies and operations (Chapin et al. 2002). This paper is focused on research questions arising from the need for new management strategies and practices. The authors emphasize the need for studies on forest growth as a key process in forest ecosystem functioning. The role of dynamics in forest growth on various time scales is highlighted with special consideration on long-term processes.

The aim is to formulate relevant research topics which should be addressed in order to provide the detailed knowledge needed for the development of appropriate management approaches. By nature these topics require integrated multidisciplinary and multi-scale approaches. Generalizations of research findings from studies on processes and the functioning of forest ecosystems will provide the most valuable knowledge basis for decision making processes on the strategic and operational management level. The suggested research topics address questions which are of relevance for the advancement of forest management all over Europe. However, due to significant differences in environmental conditions and landscape histories over Europe, regional specific issues have to be taken into account. The suggested research approaches reflect the contributions which forest growth research can add to the challenging tasks in forest sciences.

2. Managing and sustaining goods and services provided by forests

Management options today are most commonly targeted towards the end products respectively yield outcomes, e.g. maximizing the volume of high valuable timber or current net present value of forests. Sustaining environmental services provided by forests is explicitly part of the management objectives in multipurpose forestry (Weidenbach 2001; Pukkala 2003) and landscape management (Heilig 2003). However, in the past it has most often been assumed that provision of environmental services is achieved as a “by-product” of forest production without much extra effort or significant production restrictions (Rupf 1960). Hence, sustaining environmental services has rarely been regarded as a stand-alone objective of forest management. Attempts have been made to describe relations between stand characteristics and the environmental services provided by forests (Arbeitskreis Zustandserfassung und Planung 1974; Mitscherlich 1975, 1978, 1981; Spiecker 2003), but knowledge on the effects of management options on patterns and processes in forest ecosystems which are causative for specific environmental services is still rather limited, and the trade-offs between the provision of goods and services are not well understood (Johnson et al. 2002; Mårell et al. 2003).

The relevance of tree species composition, mixture and canopy structure for the effects of environmental changes on forest ecosystems and the goods and services provided by forests, their dependency on site conditions and their modification through management need to be analyzed on a broad scale. For the control and management of sustainability of forest resources under change specific scientific knowledge needs to be considerably deepened. To be of use for the formulation of new management approaches research findings need to be generalized in a comprehensive way whereas scaling issues in the temporal and spatial domain have to be considered adequately. Sensitive parameters for assessing the state of the forest ecosystems with respect to their resilience towards environmental threats need to be identified to be of use as decisive criteria for decision support in forest management.

Management of forest ecosystems has to ensure the sustained and continuous delivery of multiple interrelated and sometimes conflicting goods and services. Key hypotheses concerning principles for sustainable management of forests under change were recently summarized by Sverdrup and Stjernquist (2002). One of the basic prerequisites of an ideal forest structure in the temperate zone and beyond is that of mixed stands: it is hypothesized that increasing the proportion of mixed stands will increase sustainability by achieving greater soil fertility and biodiversity.

The severe damages in western and central European forests caused by theextreme storm events of the early and late 1990s have been recognized as indicators of major threats to sustainability and have led to an increased awareness of the susceptibility of the forest resources to disturbances (Kronauer 2000; Teuffel 2001; Weidenbach 2001). As a consequence, the importance of risks and the role of ecosystem stability have been re-evaluated. Threats also arise from anticipated environmental changes which will most probably put European forests under more severe stress (Kellomäki et al. 2000; Aber et al. 2001; Lindner and Cramer 2002).

An important impact of these discussions is that research into the issues of mixed versus pure stands and into the question of conversion from pure to mixed stands has been largely intensified in recent years (Spiecker 1999b; Hasenauer 2000; Spiecker 2000a, 2000b; Matthes and Ammer 2001; Gadow et al. 2002; Schütz 2002; Spiecker 2003; Hansen et al. 2004; Spiecker et al. 2004). However, the mechanistic test of key hypotheses is still incomplete. Our understanding of the functioning of pure and mixed forest stands and of forest stands during conversion phases with respect to the goods and services provided is not yet sufficient for decision support (Hasenauer 2000; Makkonen-Spiecker 2001; Hansen and Spiecker 2004; Spiecker et al. 2004).

3. Research approaches for the study of long-term dynamics of forest ecosystems

The emerging challenge for forest ecosystem management today is to improve our understanding of the properties and processes that allow forest ecosystems to persist in the face of environmental changes. Since temporal dynamics of forest ecosystems operate on multiple time scales, long observation periods are necessary in order to identify the key processes that govern the system's behavior at a considered time scale (Likens 1989, 1998). Responses of forest ecosystems to perturbations occur on time scales ranging from less than a year to the order of decades or even centuries, depending on the state and condition of the system and the nature, intensity and duration of the perturbation (Spiecker 1995, 1999a; Hansen and Spiecker 2004). Therefore it is important to consider that today's conditions of forest ecosystems depend on both the current environment and past events. The persistent effects of past events are maintained as legacies that can only be resolved and understood by studies which cover the characteristic time scale where these effects are expressed (Kahle and Spiecker 1996; Chapin et al. 2002).

Classes of ecological phenomena for which long-term studies are recognized as essential include:

- **slow processes**: e.g. stand dynamics, long-lasting effects of disturbances, soil development, land-use history
- **rare events or episodic phenomena**: e.g. disturbances like droughts, floods, frost, outbreaks of pests and pathogens, windstorms, reproductive patterns of trees
- **processes with high variability**: subtle processes and complex phenomena require long-term studies in order to separate pattern from noise. Subtle processes are characterized by a high-frequency variance which is large compared to the magnitude of the medium- to long-term trend (Franklin 1989). Complex ecological phenomena involve many factors interacting on different scales. Long-term studies are important in such cases to identify the relative contribution of multiple factors by obtaining an increased sample depth in time

- **multiple equilibrium states**: transient dynamics of forest ecosystems as a whole or their components between multiple equilibrium states can only be resolved by examining long-term studies.

Considering long observation periods is a key issue for generalization and up-scaling in forest ecosystem research (Bierkens et al. 2000). The straightforward approach to analyze system behavior in the long-term is the continuous observation through measurement and monitoring over extended time periods. Observational studies might be extended by implementing experimental manipulations which allow the rigorous analysis of treatment effects when based on a proper research design. However, maintaining long-term research plots and conducting continuous or repeated assessments is time consuming, costly and results will only be available after the time of observation.

Retrospective observational studies, like forest growth studies based on the stem analysis technique (Nagel and Athari 1982; Abetz 1985; Kahle et al. 2004), are valuable in providing a means for analyzing short- as well as long-term forest dynamics and for assessing the complex structure of the multi-scale relationships in forest ecosystems (Kahle and Spiecker 1996; Spiecker et al. 1996; Dhôte and Hervé 2001). By providing baseline data retrospective studies extend the time frame of long-term studies and give an opportunity for calibration (Davis 1989). They are particularly well-suited to the study of slow processes because they can include a long time span – decades to centuries – and therefore they also have great potential usefulness for recording rare events (Spiecker 1991; Olsen 1992).

Retrospective studies can provide insight into the key processes that determine the current state and long-term behavior of forest ecosystems, knowledge which is needed to examine the role of feedback mechanisms in ecosystem dynamics. Predicting long-term forest ecosystem behavior is still a challenge for process-based scenario modeling (Van Oijen et al. 2004). This is particularly true when increasingly complex models with numerous functional relations and parameters are employed (Dale 2003; Monserud 2003). The main reason for this weakness in model performance is the lack of more detailed knowledge about the role and nature of feedback mechanisms and scale dependent interactions in forest ecosystems (Pretzsch 1997; Van Oijen, in these proceedings). The longer the prediction period the more the system behavior is determined by system dynamics (Haken et al. 1995). In providing information about key-processes that govern long-term dynamics of forest ecosystems retrospective studies constitute essential components in integrated studies aiming to deepen system understanding, for the scaling-up from forest stands to landscapes (Lertzman and Fall 1998) and finally to improve the predictive power of models (Van Oijen, in these proceedings).

Due to their usually holistic perspective retrospective studies are also appropriate to contribute significantly to the implementation of ecological knowledge in forest management decision making processes (Dewar 2001; Dale 2003).

A major weakness of retrospective growth studies lies in their limited potential to infer causality. The restricted ability to assign causal relationships between the treatments and the responses is due to the fact that the approach is inherently *a posteriori*, typically implying that

- **the study units** (e.g. trees, stands, sites) might not be optimally homogeneous in their potentially relevant attributes. Additionally information on such attributes might not be available or incomplete. As the relevant time for recording such data has often passed, it is very difficult if not impossible to obtain them;
- **the treatments** might not be orthogonal to each other, producing confounding in the results. Hence, factor specific causal effects are not identifiable from the data with certainty. This drawback can be reduced by setting up the retrospective study to mimic experimental conditions (e.g. transect study, gradient study) but strict orthogonality will hardly be accomplished; and that

• the treatments are not randomly **assigned** to the study units, instead the study objects simply exist in their particular circumstances.

Consequently, empirical results from comparative observational studies need to be interpreted with adequate care. Formally taken, the most significant benefit of observational studies is that they contribute to the generation of improved research hypotheses. However, under the given research questions and circumstances, there are often no real alternatives to this approach. Before the results are generalized the hypotheses generated from observational studies need to be tested in designed experiments wherever possible.

4. Challenging research topics

4.1 Research into forest ecosystem processes and functioning

The research topics described in this subsection address the research area 'forest ecosystem processes and functioning'. Research in these topics is relevant for establishing the knowledge base needed for an environmentally and socio-economically acceptable and improved forest management that is based on ecological principles.

Long-term changes in growth and mortality in forests
In order to improve our understanding of functions and processes in forest ecosystems, changes in forest growth and tree mortality and their relations to changes in environmental conditions need to be analyzed on multiple temporal and spatial scales. Dynamics and the underlying key processes e.g. changes in site productivity, stand dynamics, effects of disturbances require special emphasis. The aim is to improve understanding, model and predict growth of trees, and stand development, and to assess ecological risks for important tree species on different sites and under different environmental and management scenarios (e.g. Lasch et al. 2002; Lindner and Cramer 2002).

Retrospective growth data originating from stem analyses conducted in important forest ecosystems in the major European regions should constitute the core of the growth data base needed for this type of analysis. The intensive stem analysis data could be complemented by data from long-term forest research plots as well as by inventory data on tree and stand growth, on the amount of wood due to mortality and/or damaged wood due to biotic or abiotic disturbances. Information on site factors has to be compiled from other assessments e.g. site classification maps. Environmental data collected in monitoring networks e.g. meteorological data, deposition data, or aggregated/modeled data like critical loads have to be compiled and standardized for uniform analysis where needed.

The European research project RECOGNITION (Relationships between recent changes in growth and nutrition of Norway spruce, Scots pine and European beech forests in Europe) (Schuck et al. 2000; Karjalainen andSchuck 2004) is an interesting example how data from different sources can be successfully analyzed in an integrated multidisciplinary and multi-scale approach to investigate key processes in forest ecosystems. Another challenging feature of this research study is the close link between an empirical correlative approach and a process-based modeling approach.

Effects of increasing stand age on growth, vitality and mortality in forests
Growth, growth responses and mortality in forest stands at different advanced developmental stages need to be analyzed with special regard to the role of tree age for ecosystem stability

and risks of abiotic and biotic disturbances. Over Europe mean age of forest stands has considerably increased during recent decades (Spiecker 1999a, 2001). The increase in stand age is often accompanied by an increase in growing stock. The aim is to assess the ecological risks which are involved in these processes. Questions which need to be addressed are: How is tree age and tree vitality linked? What is the role of tree vitality for forest ecosystem resilience? How is the adaptive potential of forests under climate change affected by tree respectively stand age?

4.2 Research into forest management and practices

The research topics described in this subsection address the research area 'forest management and practices'. Research in these topics is needed for the successful implementation of ecological principles in strategic and operational forest management and to achieve optimized provision of forest goods and services.

Forest resource management tools to improve the ecological, economical, and societal benefits from forests
New scientifically based forest resource management tools need to be developed based on a better understanding of the key patterns and processes governing the stand dynamics in pure and mixed stands. The aim is to provide management options and tools which help to optimize multi-functionality and sustainable delivery of forest products and services which are demanded by society (Carnus et al. 2001; Mårell et al. 2003; Hansen and Spiecker 2004). A special focus should be laid on the consideration of "low input forest management": effects of reduced management intensity and their ecological, economic, and societal impacts need to be analyzed at high priority.

Strategies for the conversion of forests on sites where current tree species are not adapted
Strategies for the conversion of pure even-aged forest stands of non-site adapted species into more natural forests need to be developed. At present, the conversion of mono-species mostly coniferous forests into broadleaf dominated forests is the most challenging silvicultural task in major areas in Europe (Hasenauer 2000; Klimo et al. 2000; Gardiner and Breland 2002; Hansen et al. 2004). The aim is to develop integrated management options which minimize economic and ecological risks inherent in the conversion process and to provide the techniques necessary for their implementation (Hansen and Spiecker 2002; Spiecker et al. 2004).

Management of valuable broadleaf forests
New management options, practices and techniques for improving the ecological, economic, and societal benefits of broadleaf forests will be developed. Forests with tree species such as *Acer spec., Fraxinus excelsior, Prunus avium, Alnus spec., Ulmus spec., Sorbus spec.* bear a high potential for improving economic and ecological value by application of optimized management options.

5. A network of excellence for implementation of the emerging research tasks

The suggested research topics described in the preceding chapter are of relevance all over Europe. The concept of linking research units from different disciplines of natural and social sciences that are working on relevant topics in a European wide network bears maximum potential to achieve these research challenges in a highly efficient way. Existing research activities in Europe have to be restructured and integrated in a common framework. The instruments 'Integrated Project' and 'Network of Excellence' of the 6th Framework Programme for European research offer excellent possibilities to implement these ideas.

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Sustainable Management of Slovenian Floodplain Forests at Landscape Level

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Abstract

Forests cover 57% (11 500 km^2) of Slovenia and have primarily been preserved in higher and steeper locations less suitable for agriculture, where their protection significance has been even higher. Floodplain forests on the other hand have been under severe pressure since times of the first settlement and they have been severely affected by heavy cutting of pedunculate oak and extensive regulation of watercourses over the past hundred years. In the last decades land consolidation and drainage of fields to increase crop productivity or to reclaim wetlands for cultivation have decreased structural diversity in land use. As a consequence, vitality and natural regeneration of those forests have decreased where pedunculate oak in particular is showing most severe decline.

A spatial model was used to assess the depth of the interior environment of forests and forest patches and to determine the agricultural landscapes, which have lost self-organising and regenerative properties. By using the spatial model and historical data we obtained from the forest areas and according to the data from the 18th century military maps, we can conclude the forest areas to represent the cores of the primary floodplain forests in Slovenia. In the light of further preservation of key-tree species an experiment was carried out in two bigger pedunculate-oak forest patches where critical groundwater level for efficient regeneration was defined. Not only did water regime change, but also emissions, management errors from the past and consequences of a changed water table may also play an important role in future management perspective of Slovenian floodplain forests.

Keywords: ecological monitoring; spatial model; pedunculate oak; regeneration

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Introduction

In Slovenia relatively wide spatial changes such as rapid urban growth, technological and economical development are taking place, while on the other hand areas that are considered outside of the optimal agricultural and infrastructural spectrum are exposed to the process of marginalisation. The new spatial and building regulations emphasise the policy of sustainable development of space, transcending sector way of spatial planning, and holistic treatment of space (Novak 2002), which should be organised so as to enable an economic exploitation of natural and other endogenous resources and their regeneration, as well as the preservation of environment and biotic diversity (Plut 1999). Sustainability can be achieved only if it leads to lasting mutual and synergistic benefits between people, their livelihood, their economy, and open and built-up landscapes (Naveh 1998).

With the area of 20 273 km^2 and a population of two million, Slovenia has one of the smallest proportions of agricultural land and the largest proportion of forest (57%) compared to other European countries. In the last twenty years approximately 150 000 ha of agricultural land has been left to natural succession. The trend towards fragmentation of agricultural land has not stopped and the estimate is that after the current generation of farmers some 40 000 farms in Slovenia will remain vacant. After denationalisation approximately 80% of forests will be in private ownership and the private forests will be divided among roughly 300 000 owners with an average of 2.3 ha of forest.

In Slovenia forests are of special landscape-ecological significance. Most of them are located within the area of beech (44%), fir-beech (15%) and beech-oak sites (11%) (Perko 1998). Average growing stock is at around 282 m^3/ha (Hocevar 2003) and the forests have been relatively well preserved; there are only 15% of forests where Norway spruce was extensively introduced into the sites of deciduous trees. In the vicinity of settlements and in the agricultural regions, forests were also highly influenced by litter gathering practised for centuries, which caused the deterioration of soil conditions.

Slovenian floodplain forests in particular have been under severe pressure since the times of the first settlement. Today they represent only 2% of the total forest area. These forests have been affected by heavy cutting of pedunculate oak done in the beginning of last century, extensive regulation of watercourses subjected to flooding, and unsuitable management of agricultural land. Forest decline has affected pedunculate oak, most likely because of the dryer climate, unfavourable precipitation distribution, severe hydromelioration causing changes in groundwater table, and eutrofication (Levanic 1993; Cater and Batic 1999).

Objectives of the study

Forest management planning is the main tool for the sustainable management and use of all Slovenian forests, irrespective of ownership. The forest management plans include regional plans, plans for forest management units and silvicultural plans. The main users of the forest management planning are the Slovenian Forestry Service, forest owners, forestry enterprises and various groups representing the public opinion (Boncina 2003). According to Kovac (2003) this planning lacks strong links between planning levels, efficient control and does not include conflict management and public participation. It is more and more associated with rural development and planning policies and it should become better adapted to non-forester users.

The new Slovenian act on spatial planning brings important changes and new emphases which may change the role of sectorial plans as well as the involvement of various societal interests in the comprehensive planning process (Marusic 2003). The sectorial guidelines should now be significantly different. Instead of making land use proposals, they should provide an insight into the evaluation system that each sector has developed to promote or protect its resources. A special document, called 'the landscape concept' has to integrate sectorial guidelines concerning agriculture, forestry, water management, and recreation (Marusic 2003).

Obviously, knowledge of landscape ecology has to become more integrated to the planning process to understand the interactions between forests and other land uses and to provide a basis for various management options. The aim of this paper is to present the assessment of different impacts on floodplain forests and possible interactions between forestry and other sector activities.

The concept of assessment was based on two points of landscape ecology, the landscape matrix and the hierarchical landscape structure. Following the principles of the hierarchical structure, four levels of research into forest ecosystems and cultural landscapes were connected: from the level of the region, the landscape and the ecotope to the level of individual trees. For each of these levels suitable technologies were selected. The framework for this collection of technological tools was provided by the GIS. The foundations of the spatial information system incorporated spatial data taken from maps and information sources such as Landsat TM multispectral satellite imagery, aerial photos, vegetation and geographical maps, data on surface water and water systems, data from forest survey from the forestry information system, social planning maps, data on roads and power lines and settlement data, and data from 18th century military maps from the time of the Austrian Emperor Joseph II.

At the regional level we made our assessment of the landscape structure on the basis of maps of Slovenian forests and data on the land cover in Slovenia assessed within the CORINE land cover project. For an assessment of the fragmentation of forests we used digital data derived from a map of Slovenian forests on a scale of 1:50 000. We processed the digital data using the IDRISI (Eastman 1997) and PC ARC/INFO (ESRI 1996) geographic information system. Assessment of the landscape structure was made according to the mesoregions formulated by Plut (1999). This regionalisation is based on an attempt to take into consideration physico-geographical and sociogeographical features. Based on the combination of hydrogeographical (river basins) and economico-geographical areas (influential areas of central settlements of higher rank) sustainable regions were defined. In previous studies a model of landscape types in Slovenia had been designed. More detailed descriptions of the model and landscape types can be found in Hladnik (2003). We made our assessment on the basis of a spatial model, which can be linked to the results of other experiments carried out in forest ecotopes. The land unit or an ecotope, as an expression of a landscape as a system, is a fundamental concept in landscape ecology and it provides a basis for studying topologic as well as chronological relationships (Zonneveld 1989).

A pilot experiment was carried out in north-east of Slovenia to define the importance of groundwater table in forest complex with different degrees of declining trees on deep eutric fluvisols. Every plot was represented by ten adult pedunculate oaks, with breast diameter over 35 cm. Measurements of xylem leaf pre-dawn water potential were performed every month from March to September and crown defoliation estimates at the time of full crown development (Cater and Batic 1999). Experiment confirmed differences in groundwater table and strong connection between stress and groundwater table (Cater and Batic 2000). A larger experiment was carried out in 1999 and 2000, including the same and damaged forest complex (A) in north-east of the country and (B), a new site, better supplied with water and showing less damage, located in the south-east of Slovenia. On both forest complexes new locations with low (A1) and high levels (A2) of groundwater and within each location different light conditions according to the canopy of the stand (open, minimum shelter, medium shelter, maximal shelter) were determined. Seedlings (2+2) of the site provenance were uniformly planted and fenced one year before the measurements. In each light category there were at least fifteen seedlings and none of them were treated by any chemicals. There were no natural seedlings found on the plots in open light (100% radiation). Strong deer-grazing made it impossible to find same categories of natural oak seedlings on all sites: naturally regenerated seedlings were observed only in two – light and maximal shelter categories.

Figure 1. Location of research forest ecotopes and plots in Slovenia.

Parameters were observed and measured in the spring, summer and autumn. Every aspect lasted 8–10 days. The main observed and measured parameters were

- water relations (pre-dawn xylem water potential (PWP)/adult trees, seedlings, groundwater table),
- stomatal conductance of seedlings,
- soil characteristics (Urbancic et al. 2000),
- dendrochronological analysis (Cater and Levanic 2000).

Differences between parameters were tested with AVAR and regressions for the relations between measured parameters.

Floodplain forests at the regional and landscape level

According to the phytogeographical classification of Slovenia (Zupancic and Zagar 1995) the floodplain forests are assigned to the Pre-Pannonian subsector of the Trans-Alpine sector of the Central European province and to the Southeastern Alpine Sector of the Illyrian province. Only 156 300 ha or less than 8% of the country belongs to potential natural vegetation of oak-lowland forests (Zupancic et al. 1998). The map of the potential natural vegetation had a

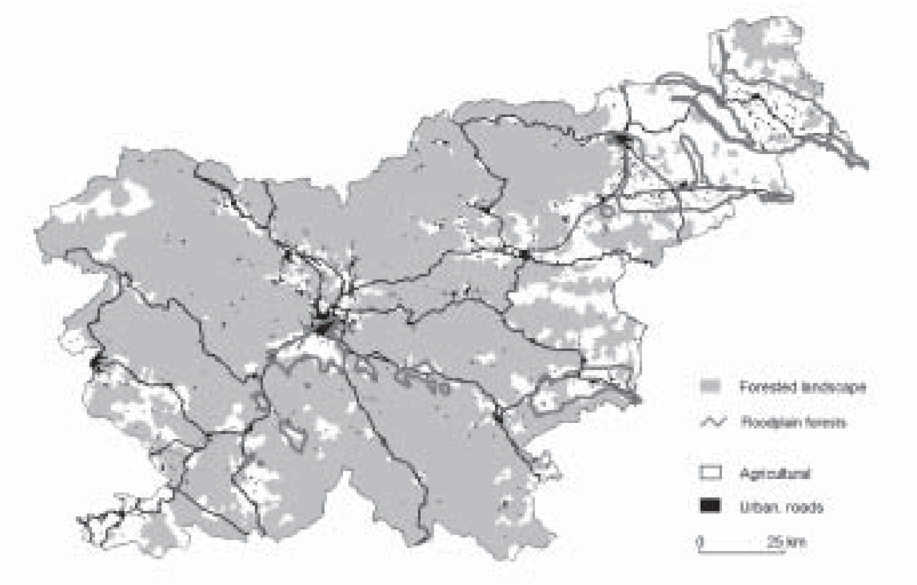

Figure 2. Landscape structure in Slovenia and the area of floodplain forests.

resolution of approximately 120 ha and a total of sixteen polygons were represented as oak-lowland forests.

Using the spatial model (Hladnik 2003) we estimated that 83% of this area belongs to agricultural and urban landscapes. In addition to intensive agricultural production, the existence of the ecological network is also threatened by the expansion of cities and infrastructure development. Further fragmentation and degradation of remnant vegetation is a consequence of these processes. The forest matrix in Slovenia is intersected by road, railway and powerline corridors. These corridors link the agricultural and urban areas and they mostly run along the plains, less than 600 m above sea level. In our estimate agricultural and urban landscapes cover 28% of Slovenia. Agricultural and urban land predominates only in six of the 25 mesoregions (Hladnik 2002). The majority of the population and of the economic activities are concentrated in urban areas and along the major infrastructure networks (Fig. 2).

According to the figures from the CORINE land cover database, arable land, artificial and urban areas account for 50% of the area belonging to the potential natural vegetation of oak lowland forests. Forest patches cover only 19% of this area. Despite small forest cover, 57% of the area of agricultural and urban land is spaced more than 150 m apart from forest edges, hedgerows and remnants of the former natural vegetation. As an open space of farmland may be defined larger complexes of cultivated land spaced over 300 m apart from forest edges, hedgerows and remnants of the former natural vegetation. Using the spatial model, such areas were determined in 26% of the area. Other farmland comprises areas with mixed agricultural use and the structure of small holdings with preserved hedgerows and remnants of the former natural vegetation.

By defining interior forest as the area within a forest patch that is at least 150 m from an edge (Hladnik 2003), in the whole area of floodplain forests only 56 forest patches with preserved interior forest environment were determined. The largest patches with interior forest environment were identified in Prekmurje (patch A) and in the Krško polje (patch B). Based on material dating back to the cartographic activities carried out between 1763 and

1787 we estimated that these patches are stable forest land which has not been subjected to agricultural use. According to the map of ameliorations in Slovenia both ecotopes were affected by changes in groundwater table. To define a critical groundwater table and light conditions which would permit successful regeneration of such forests an experiment was carried out in these forest patches of the pedunculate oak.

Results of the experiment

Water conditions

Precipitation arrangement showed differences between 1999 and 2000 and 30 years average precipitation. In year 1999 precipitation level in forest A reached 707 mm (100 mm below average) and in forest B 1300 mm (150 mm above average). In year 2000 conditions had worsened. Lack of precipitation caused severe drought in summer in both forest complexes, only 275 mm in forest A and 345 mm in forest B. Groundwater dynamics in both forests showed descend over whole vegetation period and reached minimum in autumn. The most affected one was plot A1, then the control plots B1, B2 and finally plot A2, which had the best supply with water over whole vegetation period. Pre-dawn water potential (PWP) (measurements performed every day from 3.50 a.m. to 5.30 a.m.) indicated increasing differences in values between plots towards autumn aspect (data not shown). Dependence on groundwater table was significant, especially on the site with lower groundwater (A1). Precipitation had stronger effect on the momental-daily water potential of the seedlings than groundwater dynamics, which was evident in autumn aspect in both forests, when stress was strongest. No water stress was evident in the springtime, while in the summer slight stress was detected on plot A1. In autumn 1999, moderate stress on plot A2, and severe water stress on plot A1 were present, with groundwater reaching its minimum. At the same time both plots in B forest ecotope were well supplied with water. Differences in PWP within plots with the same groundwater table showed lower negative values on open and less negative values under shelter conditions in all aspects. Conditions in the year 2000 differed more significantly between plots and also between both forest ecotopes. Stomatal conductance values showed changes when PWP values dropped below –0.8MPa to –1MPa. More evident and almost complete stomatal closure appeared below –1.6 to –1.95 MPa.

Dendrochronological analysis

Dendrochronological analysis (Cater and Levanic 2004) confirmed different growth of pedunculate oak in studied forest ecotopes. Variability of radial increment, especially after 1955, was high and tree-ring widths were hard to compare and to synchronise. Variability in growth between different radii of the same tree is in many cases higher than between the trees – a sign of a strong environmental stress.

Critical groundwater level

Values of pre-dawn water potential in different categories of seedlings were compared to the current groundwater table at the time. Critical depth for groundwater was defined from stomata closure measurements at the certain point of negative PWP values. Descend of

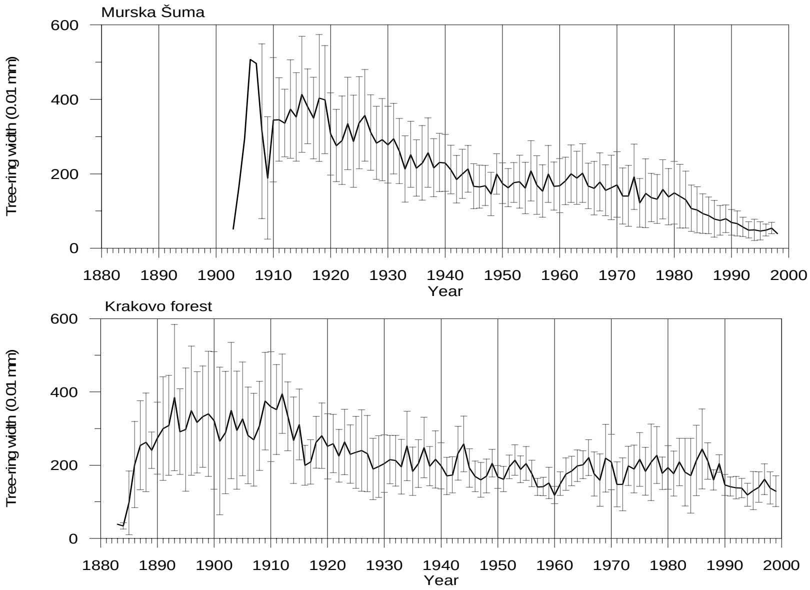

Figure 3. Variability of tree-ring widths in Murska suma (A) and in Krakovo forest (B).

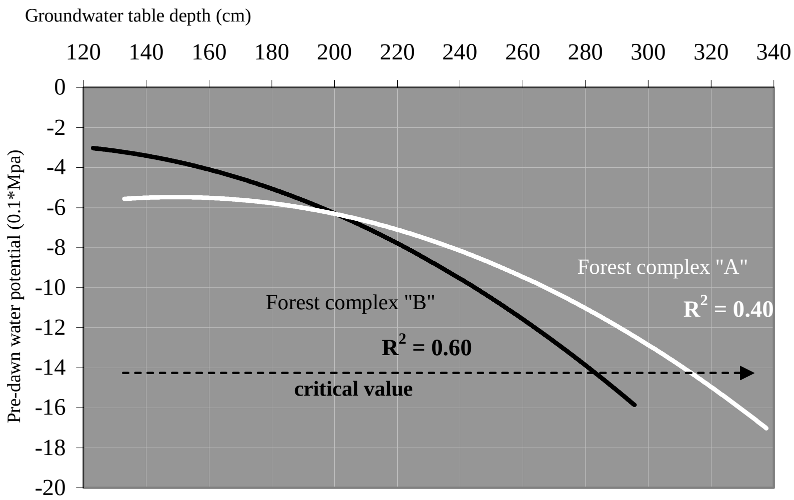

Figure 4. PWP in seedlings and groundwater for both forests.

groundwater below 260 cm induced stress, while late summer and autumn values below 300 cm with absence of precipitation caused critical descend of pre-dawn water potential values to the point of severe stomata closure, which was most evident in year 2000 (summer-autumn vegetation period).

Conclusions

Descend in groundwater table causes decrease of pre-dawn water potential in seedlings, reduces stress resistance and yield of photosynthesis and consequently stagnation in height and diameter increment. Descend of groundwater below 260–280 cm over a longer period in particular damages planted seedlings and reduces its chances of survival.

Closure of stomata in natural seedlings caused water potential below –1.80 ~ –1.95 MPa, and in planted seedlings below –1.64 ~ –1.75 MPa. Descend of groundwater table affects at most the planted seedlings under no shelter, then partially, medium and complete sheltered seedlings follow.

The complex of interactions among factors is specific to the environment. Not only changed water regime, but also emissions, management errors and consequences of a changed water table also play an important role in our oak story. Physiologically weakened trees become susceptible to many secondary, biotic factors, which would not be fatal in other circumstances. In contrast to the recovery of some other tree species, oaks have not improved in health and their decline is even increasing.

In the last decades land consolidation and drainage of fields to increase crop productivity or to reclaim wetlands for cultivation have decreased structural diversity in land use. As a consequence vitality and natural regeneration of those forests has decreased where pedunculate oak in particular is showing most severe decline. The spatial model cannot be used to forecast the development of Slovenia's agricultural landscapes and floodplain forests. It provides an overview of the landscape structure and represents a framework for more detailed research on the landscape structure or for assisting in coordination among different land users.

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Sustainable Management of Coppice Forests in Greece

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Summary

Coppices occupy an area of 8.5 mill. ha in the Mediterranean part of the European Union. They cover 65% of the total forest area of Greece. Coppice forests are considered as low output forests. The traditional management of coppices exhausts the productivity of the sites through the short rotations and grazing. The sustainable management of these forests can be obtained either as simple coppice within the same silvicultural system, or as coppice with standards (transformation), or by converting them into high forests through tending.

Using results from research carried out in the last 17 years, two aspects of the coppice silviculture are discussed in this paper: the effect of the cutting season in the vegetative reproduction process and the effect of the thinning intensity on the growth and structure of oak forests.

High mortality of stools can lead to degradation of stand structures. Planting of gaps is, therefore, often necessary as a rehabilitation measure.

Crown thinning seems to be the key for an effective silvicultural treatment of coppices under conversion.

Keywords: coppice; conversion; oak; thinning; cutting season

Introduction

When broadleaved trees are felled they will produce shoots from the cut stumps. These are known as coppice shoots, and the stump from which they grow is called a stool (Crowther and Evans 1986). Coppicing is the operation of regenerating crops in this way. Rotation times applied usually vary from 10 to 30 years (in Greece 20–30 years) depending on the region, tree species and market demands. Shorter rotations, 3–10 years, are used in energy plantations but these are not considered as forest plantations. The main products from coppice forests are pulpwood, firewood, charcoal, stakes and poles.

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In the Mediterranean part of the European Union (Greece, Italy, France, Spain and Portugal) coppices occupy an area of 8.5 mill. ha (see Table 1; Morandini 1996). The main species are oaks (*Quercus* spp.) followed by chestnut (*Castanea sativa*) and beech (*Fagus* sp). Coppice forests cover 65% of the total forest area of Greece (Table 2).

Coppice management affects biodiversity, since species connected to the clear cuts (light demanding, fast growth) are more favoured than those of the climax plant communities (shade tolerant). Moreover, many tree species do not reach maturity within the coppice rotation. Animals depending on seeds cannot survive in such forests. The more intensively these ecosystems have been exploited, the more they deviate from the climax. Intensive use (cutting, slashing and browsing) results in a lower richness of woody species and endemic taxa (Marañón et al. 1999). The often large-scale clear cuts heavily affect their aesthetic, hydrologic and protective functions.

Table 1. Area (1000 ha) of coppice forests in the Mediterranean region of the European Union (Morandini 1992, modified).

	France	Greece	Italy	Portugal	Spain	Total
Beech	81	337	528		8	837
Chestnut	54	33	647	32	24	757
Broadleaved oaks	354	1.472	965	70	590	2.726
Evergreen oaks	331		201		1.332	2.342
Other Broadleaved	42	88	907	500	282	1.834
Total	863	1.930	3.248	602	2.236	8.494

Table 2. Silvicultural Systems in Greece.

Type of Forest	1000 ha	%
High Forest	0.872	35
Coppice	1.207	48
Coppice with standards	0.433	17
Total	2.512	100

Considering that coppice forests in Greece cover 65% of the forest area or 12% of the whole country's area (13.2 mill. ha) it becomes obvious why they are less appreciated than they are in central and western Europe (Crowther and Evans 1986).

The shorter the rotations are the more their productivity decreases in long term, especially when no care is taken for their regeneration and regulation of grazing practices of livestock. In some regions, coppices were abandoned after the World War II, and they develop towards natural structures. Heavily degraded coppices show bad stem quality, low stocking, low soil coverage and high erosion rates and, therefore, are considered as low output forests. Their rehabilitation is often impossible without technical works and reforestations. The hot and dry Mediterranean climate, the shallow soil and the accelerating erosion reduce the choice of species for plantations to a few timber-producing conifers, mainly pine and cedrus species that cover the areas quickly. In any case, there is no room for plantation silviculture with the so-called fast growing species simply because the soil fertility is poor. However, the drawback of conifers is their susceptibility to fires. Taking these considerations into account, adequate planting techniques can be applied avoiding coarse terrain injuries (terracing) or extinction of native wooden vegetation (mechanically or chemically). The recovery of the potential natural vegetation can be obtained within one or two rotations of coniferous crops (100 years at least). Implementation of such a silvicultural technique may be characterized as a “close-to-nature silviculture in degraded forest ecosystems”, since there are several valuable indigenous tree species in the Mediterranean region (*Quercus*, *Fagus*, *Castanea*, *Fraxinus*, *Sorbus* spp. etc.).

Another method of rehabilitation recommended in less degraded coppices is the conversion into high forest through tending of the existing coppice stands. Appropriate age for this is after clear-cutting, i.e. at the end of the coppice rotation time. Nevertheless, the decision for conversion is often taken when stands have reached their coppice rotation (20–30 years). Main drawback of this method is that oaks do not respond to thinnings when they have not been adequately tended. They react to the first conversion thinning by building epicormics. Moreover, whole branches, tree tops or even whole trees die.

Dafis (1966, 1969) and Stamou (1980) recommend conversion into high forest by tending in good site qualities or conversion by introducing conifers (enrichment) in poor ones. Serrada et al. (1996) recommend conversion of *Quercus ilex*, *Quercus faginea* and *Quercus pyrenaica* coppices into high forest without considering site criteria. The abandonment of mountainous areas after the World War II had as a consequence the abandonment of any management of coppice forests. This trend has led to an improvement of the stand structures and the ecological conditions in general. On the other hand it has led to increased wildfire hazard (Morandini 1998). Abandonment is not necessarily bad, but “no management” may have adverse results on the coppice forests in long term. Since conversion is not only a financial but also a social and environmental problem as well forest owners, policy makers and the public will decide about their future management options.

The aim of this paper is to present results from two experiments, which are concerned with two important issues of the management of *Quercus frainetto* coppice forests in Greece.

Materials and methods

Effect of cutting season on the resprouting of stools of *Quercus frainetto*

The regeneration of the coppice forests is directly connected to the cut of the mother stand. The resprouting at each stool varies in time, growth, number and vigour of the shoots according to the season of the cut. This is important for the first years, and affects the stand stability and quality for the rest of the rotation time. Fewer sprouts per stool grow faster than numerous sprouts. Shoots from late summer and autumn cuts can be damaged by early frosts. On the other hand, inappropriate season of cut can cause even the death of stools, and consequently bad stand structures.

The experimental area is located in the public forest of Sochos (Prefecture of Thessaloniki, Northern Greece). The elevation is 600 m and the bedrock is two mica gneiss (gneiss with biotite and muscovite). The vegetation zone is *Quercetalia pubescentis*, occupies the zone between *Quercetalia ilicis* and *Fagetalia*. The climate of this area is a transition from the typical Mediterranean climate to the continental one, the mean annual precipitation is 585 mm and mean annual temperature 12.1 °C. Frost and snow can cover the soil for 1–2 months each year, while the drought period is 1.5–2.5 months.

The effect of cutting season on resprouting and growth of the young shoots has been investigated in a block design with six cutting dates throughout the year (April, June, August, October, December, February) and three replications. Each treatment plot was 500 m^2 (6x3x500 = 9000 m^2).

Measurements have been carried out three times in the first year and once per year in 1996, 1997, 1998 and 2000. Measurements included the origin of buds (dormant or adventitious), date of first sprouting, height and diameter at 5 cm above stump or soil level, the inclination from the vertical, the story within shoots of the stump and possible damages from frosts, insects and fungi.

Effect of silvicultural treatments on growth of young *Quercus frainetto* coppice stands

A randomised block design has been established in 1986 in a 9 years old oak stand (*Qu. frainetto*) in Zangliveri/Thessaloniki, Greece. The vegetation zone and climate are similar to Sochos forest but the bedrock is Quartzite with Phyllite.

Three blocks (replications) and four treatments (50x60 m) were applied: control, light, moderate and heavy treatment. Each plot has a core area of 1200 m² (30x40 m), surrounded by 10 m buffer zones. The first preparation tending has been applied in 1986 (age 9 y.), and about 30% of basal area has been removed. In 1994, the thinning intensities mentioned above have been applied. Crown thinning with positive selection was applied (German term: "Auslesedurchforstung"). All trees were tagged and assigned in three silvicultural categories the candidates (trees to be favoured), the competitors (trees to be removed) and the neutrals (all other trees), and they have been measured by BHD, total height, height of green crown and stem quality of candidate trees. As a rule of thumb, candidates were chosen in a spacing of 3x3 m (900/ha). After choosing them, one, two or three competitors were cut according to the thinning intensity. Thinning intensity can be defined as the ratio of number, basal area or volume of trees removed in relation to the total number, basal area or volume before thinning of the same plot or of the control plot.

Results and discussion

Research related to the species *Quercus frainetto* is very restricted. Moreover, specific problems, like the effect of the cutting season and conversion thinnings do not exist at all. In the frame of the MEDCOP project research efforts have initiated dealing with oak species and chestnut.

Effect of cutting season

- Resprouting started within two months after cut. Even stumps cut in October resprouted before the end of the year. In all treatments, resprouting initiated in the same year but also in the second and in a few cases even in the third year. Stools continued to build new sprouts in the second year and, a limited number, even in the third year. Resprouting stopped after the third year.
- Cutting dates before leaves initiation (December–April) resulted in maximum sprout numbers in the first year, and a decreasing number of new sprouts in the following years.
- Stump mortality does significantly reduce yield per Hectar by lowering stocking, and replacement should be carried out. Our investigations have shown following results: stool

mortality was low in the August cut (6%), middle from October to February (12–20%) and high in the spring cuts from April to June (28–30%). Ciancio et al (1996) have found that the cutting season did not have any effect on stool mortality and the capacity of regrowth in holm oak (*Quercus ilex*) in Italy. Carvalho and Loureiro (1996) investigated *Quercus pyrenaica* in Portugal and found similar results with holm oak. The different results could be due not only to different species, but also to the high annual precipitation that exceeds 1100 mm and to the very short drought period in relation to the Greek site. In sweet chestnut (*Castanea sativa*) coppices, in United Kingdom, typically 5% of stumps die at each cutting but in lime (*Tilia sp*.) the figure is only 1–2% (Crowther, Evans, 1986).

- Total sprout mortality in the first six years was 77–90%. Already, three years after cut, only three shoots dominated and made up the overstory, while three times more sprouts remained in the under and middlestory as a result of competition within sprouts of the same stool (within stool competition).

Thinning

Young coppice stands have a different structure than stands from seed regeneration. Trees grow as clumps and use the same root system, while trees of seed origin have their own root. When a sprout is cut, adjacent sprouts are also favoured and get not only more light but also more nutrients from the common root system. These features must be taken into consideration when tending and thinning such young stands. On the other hand, removal of one or two sprouts favours predominant sprouts to grow much faster than co dominant or dominated ones. As sprouts grow in height and diameter, stem quality may be inferior owing to the

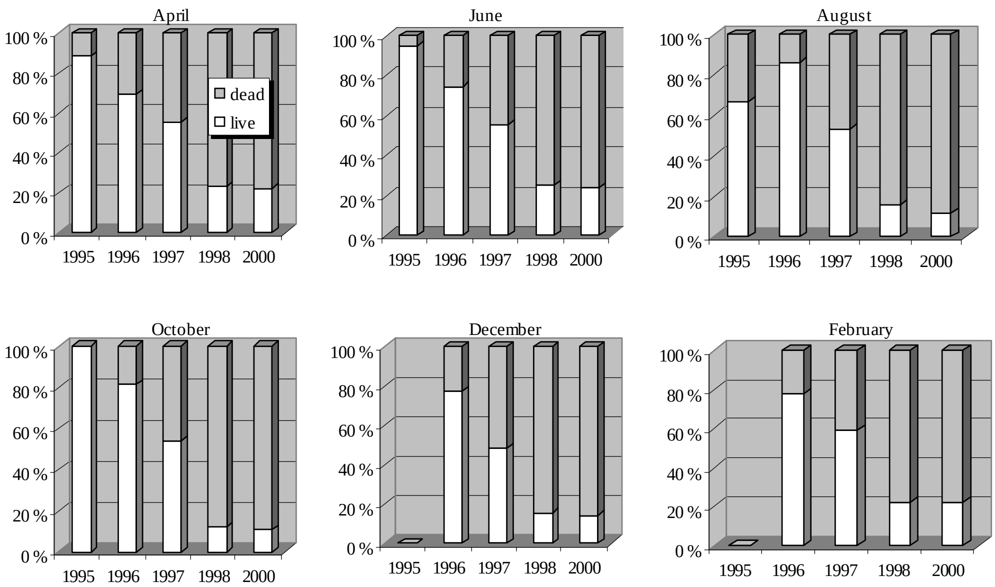

Figure 1. Dynamics of shoots among different cutting dates. Survival rate six years after clear-cut varies between 10% in the October cut and 23% in the April cut.

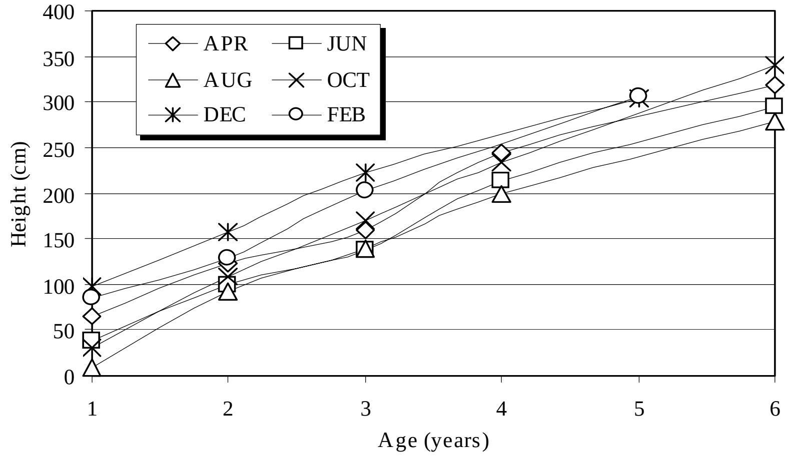

Figure 2. Development of mean heights of the three dominant shoots of each stool for each treatment (cutting season).

curved or swept butt (Crowther 1986; Evans 1986). For these reasons, coppice stands must be thinned as early as possible. Removing trees from the overstory that heavily affect the candidates is the best way to improve the young stands both in quality and stability. The results below are the description of the dendrometric data before and after the first thinning.

- The majority of removals in the experimental plots belonged to the upper diameter classes (Figure 3).
- Thinning intensity expressed as percentage of number of removed trees was always lower (13, 21 & 27%) than the corresponding intensity, expressed as percentage of the basal area of removed trees (17, 28 & 34%, Figure 5)
- The ratio "removals/candidates" (%R/C), as a measure of how intensively candidates are released, varied from 0,8 for the light thinning, to 1.2 for the moderate and to 1.7 for the heavy thinning (Table 4). These rather conservative thinning intensities are justified for oak, which is a slowly growing species and does not promptly react to crown release.
- The mean diameters of candidates and removals were bigger than the diameters of the neutrals (Tables 4 and 5), because the choice of candidates is made from the strongest shoots of each stool (Figure 4). The competitors were chosen from the dominant or co-dominant stand layer as well, while neutrals, comprising 40–60% of basal area, were thinner than the first two categories, because this category included all dominated trees.

Cañellas et al. (1996) conclude that tending of *Quercus faginea* stands under conversion should start early and thinnings should be moderate or heavy, and the light thinning did not have any effect on growth.

However, direct comparison between the two experiments is not easy, because in the moderate thinning they removed 42% and in the heavy thinning 57% of stems compared to the light thinning. Moreover, it was a thinning from below, and all stems were removed except the remaining ones which were supposed to be the candidates or plus trees.

As far as the thinning intensity is concerned, too heavy thinnings reducing considerably the standing volume, have a negative impact on the diameter increment (Amorini et al. 1996).

Table 3. Biometric data of the experimental plots per ha (age 17 y) before thinning

PLOT	Tree density (N/Ha) Overstory	Understory	Sum	D (cm)	H (m)	G (m^2/Ha)
Control	6.092	1.786	7.878	4.9	5.1	12.5
Light	5.497	2.208	7.706	5.0	5.3	11.9
Moderate	5.211	1.930	7.141	5.3	5.5	12.6
Heavy	4.889	2.233	7.122	5.4	5.6	11.8

Table 4. Silvicultural attributes of the trees of the experimental plots (values per ha).

	Candidates				Neutrals				Removals				
PLOT	N	D	H	G	N	D	H	G	N	D	H	G	%R/C
Control					6.092	4.9	5.1	12.6					
Light	831	6.5	5.6	2.9	3.967	4.6	4.6	7.1	700	5.7	5.4	2.0	0.8
Moderate	956	6.5	5.7	3.3	3.153	4.6	4.7	5.8	1.103	6.2	5.5	3.6	1.2
Heavy	814	6.5	5.8	2.7	2.725	4.7	5.0	5.0	1.350	6.2	5.6	4.1	1.7

Table 5. Biometric data after thinning.

	N	D	H	G
Control	6.019	4.9	5.1	12.4
Light	4.797	4.9	5.2	9.9
Moderate	4.108	5.1	5.4	9.0
Heavy	3.539	5.1	5.5	7.8

Where: N is number of stems/ha
D is breast height diameter (cm)
H is tree height (m)
G is basal area (m^2/ha)
%R/C is the ratio Removals/Candidates in %

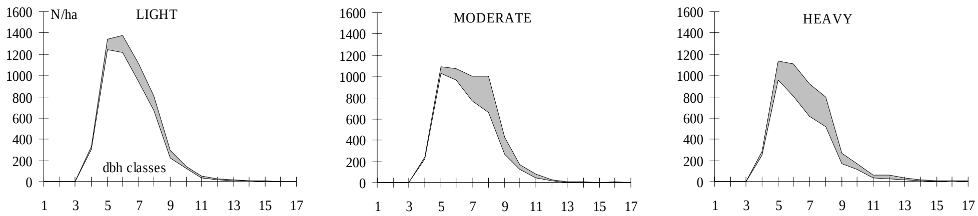

Figure 3. Distribution of diameters (cm) before and after thinning 1994. The right skewed hatched area illustrates the intensity of a crown thinning.

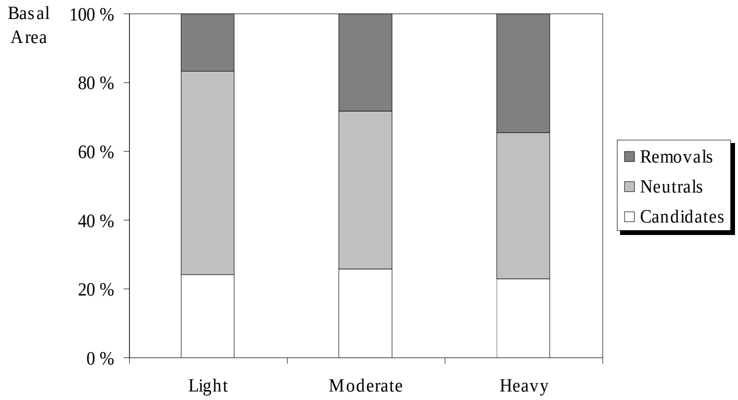

Figure 4. Diagram illustrating the relationships between numbers of trees and basal area per ha as mean values of the three treatments (light, moderate and heavy) for each silvicultural category. Percentages of basal area of candidates and removals are higher than the corresponding percentages of numbers of trees. In contrast, higher percentages of number of neutral trees show a relatively smaller percentage of basal area, as a result of the thinning concept (crown thinning)

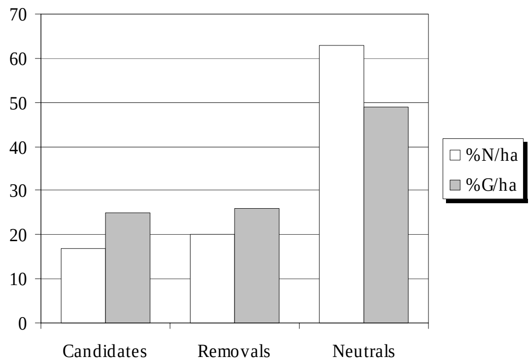

Figure 5. Distribution of basal area of candidates, neutrals and removals in each thinning intensity. Candidates comprise 25% of the total basal area among all treatment intensities. The basal area removed was 17, 28 and 34% in the light, moderate and heavy thinning respectively.

Conclusions

Sustainable management of coppice forests includes three main options: to keep on the same system, to convert into coppice with standards or to convert into high forest through tending. Silvicultural decisions should take into consideration the following principles:

Simple coppice: Resprouting and fast first-growth are desired. Thus, cuts must be carried out from October to March, in order to produce vigorous sprouts. The manager has to make observations up to three years following the clear-cut and decide on possible additional artificial regeneration in case that resprouting, root suckers and seed regeneration cannot cover the area. In order to avoid further site degradation, longer rotations (40–50 years) must be preferred. Further prolongation of rotation times is not recommended because tree density would decrease and the gaps after each final cut would be open for long time. The most appropriate density is considered to be 1000–1500 stools per ha.

Coppice with standards: Generally, it is not recommended. If applied for special reasons, the choice of the standards must follow specific criteria, i.e. length, shape and vitality of the crowns, dead branches and stem quality. They should be evenly spaced and adequately prepared through thinnings.

Conversion into high forest: The conversion regime has to be defined in detail. This includes time of commencement, thinning schedule, type and intensity of thinnings and possible rotation length. Conversion treatments should start from the better sites and as early as possible, preferably after the final cut. Silvicultural treatments must take place in unfavourable season for resprouting in order to eliminate unnecessary competition. Oak crowns do not react to thinnings if they are not prepared through early and frequent interventions. Thus selected trees must be vigorous, avoiding uniformity in stand structure. Positive selection is the most efficient technique for achieving the best stand structures.

Crown thinning results in well-structured stands with a healthy overstory, consisting of well-formed, long crowned trees, while crowns of the middlestory protect their stems. Dominated trees should not be subject of thinnings. They will build a dynamic middle and under story that will fill in any gaps created by the interventions. The crowns of well-developed dominant trees react promptly to any release. In all cases, forest managers have to take into consideration that coppice forests need to rehabilitate. Grazing must be therefore regulated for the time needed and the minimum timber dimensions that can be removed from the forest must be defined.

In contrast, in uncultivated stands, by the age 25–30 years, up to two bad shaped stems remain at each stool as a result of the competition between trees from adjacent stools. It is therefore too late to start converting them.

Enrichment with conifers must take care for the temporal co-existence with the natural tree species. Planting techniques must avoid injuries of landscape and soils. In the first rotation time, the planted species should be favoured because of the competition from the shoots. In the next rotations, more and more native tree species will replace the introduced ones, either as a result of natural or a combination of natural and artificial regeneration. Native broadleaved species must be regenerated with seeds, which implies rotations longer than 50 to 70 years.

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The Effects of Silviculture on the Structure in Mature Scots Pine Stands

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Abstract

The aim of this paper is to assess the influence of silvicultural treatments on the structural diversity of Scots pine stands in Spain. Two mature Scots pine stands (80–100 years) with different origin and silviculture applied were chosen to develop the study. Spatial pattern, diameter distribution, diameter, height and crown length differentiation, foliage height diversity (FHD) and coarse woody debris (CWD) are compared in both stands. More structural diversity was found both in the spatial pattern and CWD of the forest where the shelterwood system was applied over a longer period, although the forest where silviculture is more intensive had greater FHD. Different aspects of silviculture influence different aspects of stand structure, in a rather logical way, while other structure characteristics remain constant.

Keywords: silviculture; stand structure; diversity; Scots pine

1. Introduction

Biodiversity has become one of the main topics in forest management and conservation, since the earth Summit in Río de Janeiro in 1992. However, biodiversity is an abstract and often complex concept which should be specifically defined according to the context in which the term is used (Parviainen et al. 1994). The term can be used at several biological levels from landscape diversity to genetic diversity. Moreover, each biodiversity level covers three aspects: composition, structure and function.

The structural diversity of the forest ecosystem is an important component, since it can be partially modified by silvicultural treatments. Forest structure seems related to the habitat of many plants and animals (Mac Arthur and Mac Arthur 1961; Degraaf et al. 1998; Ferris-Kaan et al. 1998), and can be used joint to other factors, such as the site quality, the successional

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gradient or the tree species composition (Wiens 1989; Bersier and Meyer 1994; Pitkänen 2000) as indicator of biodiversity (Kuuluvainen et al. 1996). Nevertheless, complex forest structures do not always mean high diversity and often the opposite may be found (Hunter 1990). Stand structure is also directly related to many fundamental aspects of the forest, like stability, production, soil protection, landscape, etc. Therefore, an adequate knowledge of the forest structure and its dynamics is necessary to guarantee sustainable management. Usually, the forest structure is characterized by considering four aspects: spatial pattern, mixture and diversity of species, horizontal and vertical differentiation and coarse woody debris (Gadow and Hui 1999; Lähde et al. 1999; Weber 2000).

Management objectives for *Pinus sylvestris* L. forests in Spain are, in general, to guarantee the survival and stability of the stand, to optimise multiple use, and to achieve sustainability of yield, including indirect benefits like soil protection, recreation, etc. These objectives vary according to the ecological characteristics of the stands and the social needs of the area. In the same way, the silvicultural treatments must be adapted to these conditions and objectives.

The Scots pine is one of the most studied forest species in Spain, including extensive studies regards its ecology, genetics, stability, production, etc., and the effects of silviculture on these aspects (Cañellas et al. 2000; Montero et al. 2001). However, some fundamental aspects for sustainable management are poorly documented, such as, prevention of global warming and CO_2 fixation, maintenance and increase of biodiversity, conservation of genetic diversity, etc. (Bastien and Alía 2000; Caparrós et al. 2001; Río et al. 2003).

Given the necessity for research in these areas, this study was developed to increase our knowledge on the influence of silvicultural treatments on the current structural diversity of Scots pine stands in Spain. Firstly, the structural diversity of both forests will be described and secondly, the relationship between this diversity and the silvicultural treatments applied will be analysed.

2. Material and methods

2.1 Study site

Two natural and even aged Scots pine stands were chosen in the Central mountain range of Spain to carry out the study: Navafría (4436 ha) and Valsaín (7627 ha) forests. Both forests have similar ecological conditions (altitude range from 1200 m to up to 2200 m; north facing; acid soils) but different silvicultural systems have been applied over the last century. In the Navafría forest the shelterwood system was used, being the trees removed in a few heavy fellings and the shelter-phase 20 years. If natural regeneration has no success, the fellings were followed by soil preparation and seedling or plantation. An intensive thinning regime is also applied. In contrast, the shelterwood system applied in the Valsaín forest was much more gradual and regeneration was allowed to take place naturally, leading to a long regeneration period (often over 40 years). The thinning regime was less intensive than the one applied in Navafría. The management plans started in 1898 in Navafría and 1889 in Valsaín, and the rotation is 100 and 120 years respectively.

The use of large plots is common in recent studies focused on forest structure assessment (Ferris-Kaan et al. 1998; Aguirre et al. 2003; Kint 2003). It allows the precise characterization of the stand structure, although it often means to give up the replication of the trial. A plot of 0.5 ha were established in the 80–100-year aged block of each forest to compare stand structure diversity of mature stands in both forests. Places were the silviculture applied is representative of the silvicultural system used throughout the forest were chosen: in

Table 1. Main characteristics of the Valsaín and Navafría plots.

Plot	Trees/ha	G	Mean Age	Dg (min/max)	Ht (min/max)
Navafría	183	24.09	101	40.94 (15.4/54.4)	20.53 (16.4/25.1)
Valsaín	275	26.86	105	35.30 (28.8/66.8)	22.16 (16.6/29.9)

G: basal area (m^2/ha); Dg: Mean diameter (cm); min: minimum dbh; max: max dbh; Ht: mean total height (m); min Ht: minimum total height; max Ht: maximum total heigth.

Table 2. Characteristics of wood and subplots size for each fraction of coarse woody debris (CWD) considered.

CWD type	Size	Subplot size (m^2)	Subplots/plot
Dead standing trees	∅≥10 cm	5000	1
	10 cm≥∅>5 cm	100	4
Stumps	∅≥30 cm	5000	1
	30 cm>∅≥5 cm	100	4
Fallen logs and branches	∅≥10 cm	5000	1
	10 cm>∅≥5 cm	100	4
	5 cm>∅	25	4

Navafría the origin of the stand is a plantation made about 1910, whereas in Valsaín the natural regeneration has established. Plots were located similar altitudes (1400 m in Navafría and 1200 m in Valsaín) and with similar site qualities (23 m in Navafría and 26 m in Valsaín). Stand density is still high at this stage of development with an absence of regeneration or shrubs. Consequently, the diversity of the stand has been analysed by considering the structure formed by the trees and the abundance of different types of coarse woody debris (CWD). Diameter at breast height (cm), crown diameter (m), total height (m), crown length (cm) and position of all trees were measured in each plot. Age was taken using increment borer in a random sample of 30 trees with the same diametric distribution than the stand within the plot. Table 1 shows the main characteristics of the plots (number of trees, basal area, mean, age, mean diameter and height).

The number and size of subplots used to measure CWD vary according to the CWD types from 25 m^2 to 0.5 ha (Table 2).

2.2 Stand evolution

From the compilation of the management plans and their successive revisions, we have obtained data regarding the number of trees per diameter class, information about cutting, seedlings, natural regeneration and other silvicultural treatments for these forest compartments over the last century.

From the available data it is not possible to ascertain the diametric distribution prior to the shelterwood cuttings, but the description of the previous stand tells us that there was advanced regeneration and that the upper strata did not form a closed canopy, but rather a

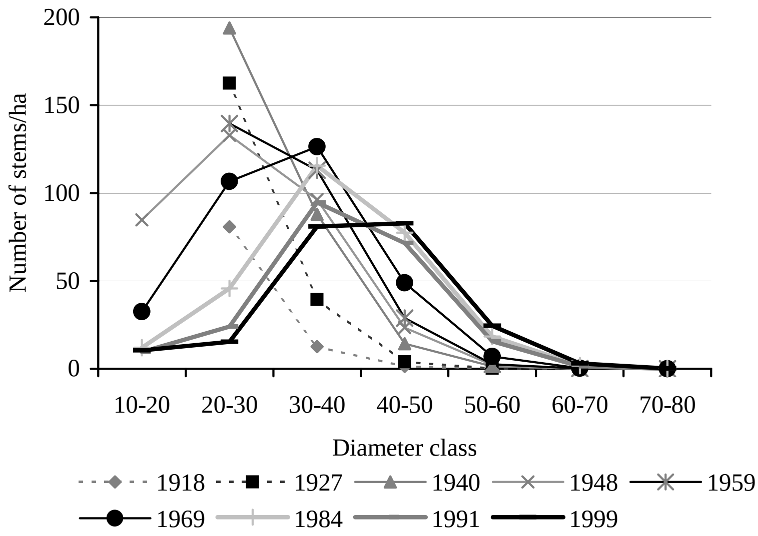

Figure 1. Variation of diameter distribution in block 1A1 of Pinar de Navafría between 1918 and 1999. There were no data about stems in the 10–20 diameter class in the inventories of 1918, 1927, 1940 and 1959.

sparse canopy of older trees. In the Navafría forest compartment (Figure 1), the release of upper strata was carried out over a 20-year period and regeneration was established throughout the forest compartment. Natural regeneration had to be helped with plantation in some places. This promoted a dense regular stand. Since then, an intensive thinning regime has led to a decrease in stand density. As stated above, the stand within the experimental plot comes from plantation.

In Valsaín things happened differently: the Management plan (1889) was dropped eight years later in favour of an individual tree selection system (Donés 1994). From 1948 to 1965 felling followed the initial Plan with some delay (Montes et al. 2003). Figure 2 shows how regeneration established itself since 1941 to 1965, simultaneously to the development of the stand established before. Since 1988 the group selection system has been applied in the forest. In Figure 2, stem distribution in 1988 and 1998 is related to part of the initial forest compartment because due to the high heterogeneity within the forest compartment, it was divided, and the other part started regenerating in 1988. The experimental plot was located in a 90 aged stand within the forest compartment. It can be noticed that the diameter distribution at forest compartment level (Figure 2) is skimmed to the left in comparison with the diameter distribution within the plot (Table 1 and Figure 5), because the mixture of different aged stands due to the long duration of the regeneration process for the whole forest compartment. This increases the structure heterogeneity at scales larger than the plot, which this study is focused.

2.3 Plot structure characterisation

The structural diversity has been analysed by considering two aspects: the structure formed by living trees and the abundance of different types of coarse woody debris (CWD). The spatial

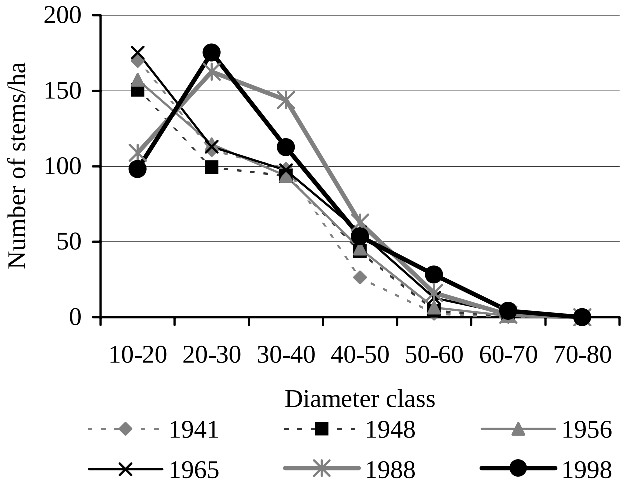

Figure 2. Variation of diameter distribution in block 1A1 of Pinar de Valsaín between 1941 and 1998.

pattern, diameter distribution, microstructure, horizontal and vertical differentiation and foliage height diversity (FHD) have been used to characterise the structure related to living trees. In computing distance dependent indices, problems arose with those trees considered border trees (trees with neighbours out of the plot). Those trees which were nearer to the boundary of the plot than to the farthest neighbour being considered for each index were excluded. The CWD has been characterised through volume of dead trees, stumps, logs and snags and weight of litter.

2.3.1 Spatial pattern

The *k(d)* Ripley's function has been used to study the spatial pattern of trees in each plot (Ripley 1981). In this function the expected number of trees within a radius d with a random distribution is compared to the empirical function $\lambda K(d)$:

$$\lambda K(d) = \sum_{i=1}^{n} \sum_{j=1}^{n} \frac{\delta_{ij}(d)}{n}, i \neq j, \qquad \delta_{ij}(d) \begin{bmatrix} 1 & if\ d_{ij} \leq d \\ 0 & if\ d_{ij} > d \end{bmatrix} \tag{1}$$

where λ is the density, d_{ij} distance from *i* tree to *j* tree, and *n* the number of trees in an area with radius *d*. The *k* value is compared to the expected value of a Poisson distribution obtained through 99 Monte Carlo simulations. In order to represent graphically the Ripley's function, the transformed function *L*(d) has been used:

$$\hat{L}(d) = \sqrt{\frac{\hat{K}(d)}{\pi}} - d \tag{2}$$

Gadow's angle uniformity index has been used to assess the symmetry of the spatial distribution

$$I_G = \frac{1}{n} \cdot \sum z_{ij} \qquad z_{ij}(d) \begin{bmatrix} 1 & if\ w_{ij} \leq w \\ 0 & if\ w_{ij} > w \end{bmatrix} \tag{3}$$

where n is the number of neighbours considered, w_{ij} is the angle formed by lines from the tree of reference to i and j neighbours and w is the ratio of 360º to n.

2.3.2 Diameter distribution and size differentiation

Diameter distribution has been analysed within the plot. In order to characterise the microstructure situation the vertical and horizontal differentiation are estimated with the Gadow's differentiation index (Gadow, 1993):

$$TD3 = \frac{1}{N} \sum_{i=1}^{N} TDn_i \qquad TD3_i = \frac{1}{3} \sum_{j=1}^{3} \left(1 - \frac{x_{\min}}{x_{\max}}\right)_j \tag{4}$$

where TD3 is the mean differentiation calculated with 3 neighbours, N the number of trees analysed, $TD3_i$ the differentiation index for tree i calculated with 3 neighbours and x_{min} and x_{max} are the smallest and the largest value of the feature analysed (diameter, height or crown length) when comparing reference tree i and its 3 nearest neighbours.

2.3.3 Foliage height diversity

Foliage height diversity (*FHD*) is estimated using the Shannon index (*H'*) (MacArthur and MacArthur, 1961).

$$H' = -\sum p_i \cdot \ln(p_i) \tag{5}$$

where p_i is the relative abundance of foliage in strata i. To estimate the relative abundance of foliage the crown has been assimilated to an ellipsoid with the y axis equal to crown length and the x axis equal to crown diameter. The ellipsoid volume has been calculated within height strata through the following integral:

$$\begin{aligned} V_i &= \int_{h_{i1}}^{h_{i2}} \pi \cdot r^2 dh \\ r^2 &= (d_c/2)^2 - \frac{(d_c/2)^2 \cdot [h - (h - l_c/2)]^2}{h_i^2} \end{aligned} \tag{6}$$

Table 3. Rot classes of coarse woody debris.

Rot class	Stage of decomposition
I	Freshly dead, at most one year old
II	Wood hard, bark partly loose but >50% remaining
III	Wood hard or soft in the surface, <50% of the bark remaining
IV	Wood soft in the surface or throughout
V	Wood soft throughout or with a hard core only, the outer surface hard to distinguish or completely or partly covered by forest floor mosses

where V_i is the crown volume for the tree in the *i* strata (from h_{i1} height to h_{i2}), *r* is the radius of the cross section of the ellipsoid at *h*, d_c is the crown diameter, l_c is the crown length and h_t is the total height of the tree. Lower strata comprises from ground to 2 m height, second from 2 to 5 m, third from 5 to 15 m and upper strata above 15 m height. These strata are the same as those used by Ferris-Kaan et al. (1998) to assess vertical structure in managed Scots pine forests in Britain.

2.3.4 Coarse woody debris

In order to study CWD 8 classes of elements (Table 2) and 5 rot classes (Table 3) have been established (Siitonen 2000). Volume of branches and logs (estimated from length and mean diameter) and number of stumps by size class and rot class is used to characterised the CWD.

3. Results and Discussion

3.1 Spatial pattern

Figure 3 represents the distribution of the trees in the two plots through the crown projection. Spatial patterns of trees studied through the transformation $\hat{L}(d)$ of Ripley's K are presented in Figure 4. Light lines indicate 90% confidence interval boundaries of function $\hat{L}(d)$ of a Poisson distribution. When $\hat{L}(d)$ function for real distribution of the trees (bold line) falls above the confidence interval upper boundary the distribution is clustered; if it falls under the lower boundary the distribution is regular. The spatial patterns are similar in both plots for smaller distances (<10 m) with a tendency towards regularity. Up to 11 m the spatial pattern in the Valsaín forest is significantly aggregated, while in the Navafría forest it is random in the studied scales. These results show the possible effect of the natural regeneration way on stand structure in Valsaín, with regular and random patterns on a small scale due to competition and remaining a cluster pattern on larger scales. This aggregated pattern hasn't been found in Navafría the stand comes from a plantation. It is interesting to note that the effects of the regeneration way continue up to an age of 80–100 years in the Valsaín plot, because the spatial pattern tends towards regularity through the phases of stand dynamic (Moeur 1997). The variation of the spatial pattern will be studied by comparing the results of the 6 age classes in each forest.

Gadow's uniform angle index, calculated with three neighbours is 0.65 for the Navafría plot and 0.63 for Valsaín, which shows that very similar pattern is found in both forests at microstructure level. It is likely that microstructure is more related to species growth and competition, and that the pattern at larger scales is more influenced by the silviculture system.

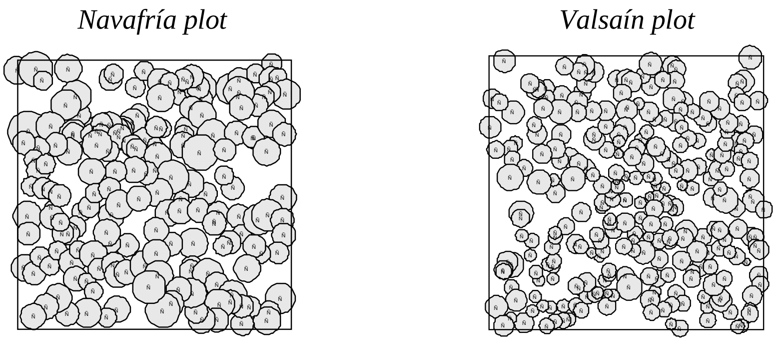

Figure 3. Distribution of trees represented by the crown projection of Navafría and Valsaín plots.

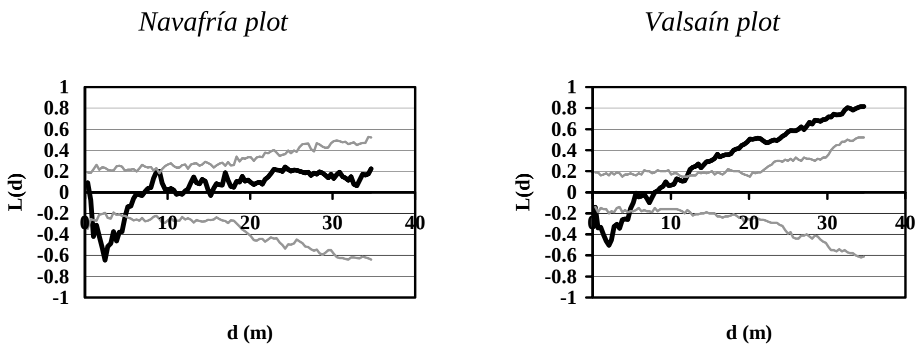

Figure 4. Analysis of spatial pattern using the transformation *L(d)* of Ripley's function *K(D)* in Navafría and Valsaín plots. Bold lines: K function value for the real distribution of the trees; Grey lines: 90% confidence interval boundaries of *L(d)* of a Poisson distribution.

3.2 Size spatial arrangement

Figure 5 shows the diameter distribution within the plot. In both plots the diameter distribution corresponds to a single-cohort stand, but in Valsaín it is less regular: Coefficient of variation of diameter (CvDbh) is equal to 19.6%. In Navafría CvDbh value is 15.1%.

Another important aspect of structural diversity is the degree of mixture of different sizes or differentiation. Gadow's index shows the differentiation of the microstructure because it is calculated with the 3 nearest neighbours. Mean values of Gadow's indices for Navafría and Valsaín are 0.14 and 0.16 calculated for diameter, 0.07 and 0.07 for total height and 0.24 and 0.26 for crown length. These low values for Gadow's indices correspond to a low level of differentiation according to the scales proposed by Füldner (1995) and Aguirre et al. (1998), which is typical of mature stages of single-cohort stands. The regularisation is caused by self-thinning and favoured by the application of low thinning. Differences between plots in the distribution of values for Gadow's index are greater when the index is calculated using crown lengths (Figure 6), indicating that the plot of Valsaín shows a lower degree of regularisation with more social differentiation.

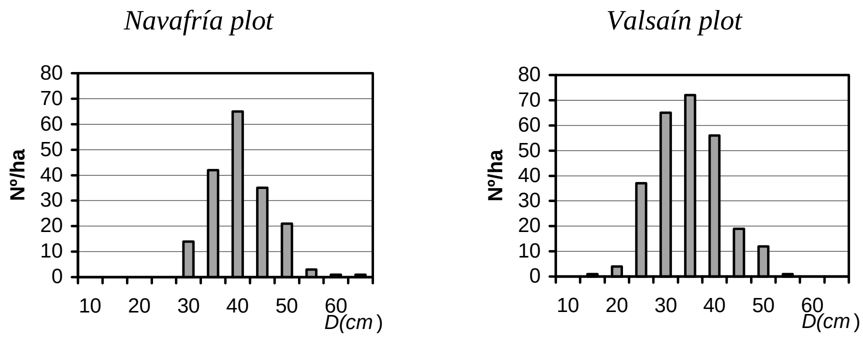

Figure 5. Diameter distribution of the Navafría and Valsaín plots.

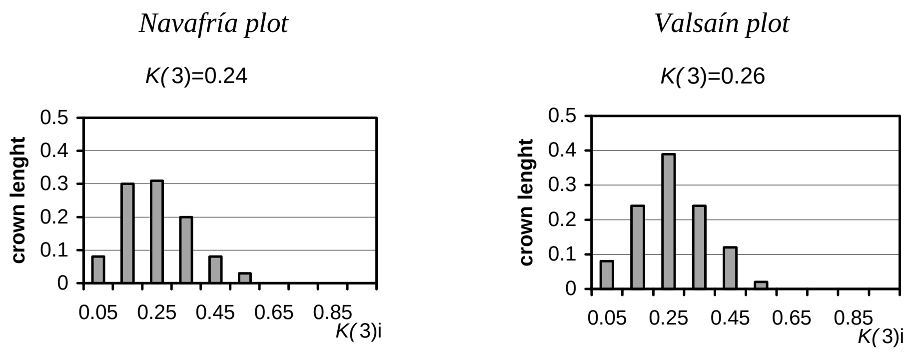

Figure 6. Mean value K(3) and distribution of Gadow's differentiation index (1993) calculated for crown length with the 3 nearest neighbours in Navafría and Valsaín plots.

FHD gives very low values since both are one-storied dense stands without shrub strata. The FHD value is 0.43 in Navafría, and 0.19 in Valsaín. The thinning regime has lead to lower density at the Navafría plot, allowing the remaining trees a greater crown development. The canopy occupies a great part of the strata from 5 to 15 m and the strata above 15 m height. In Valsaín the crowns are less developed because the high density and the canopy occupy mainly the upper strata. The results obtained in other reports (Barbour et al. 1997) suggested that thinning could accelerate the development of some aspects of stand structure found in late seral stage forests. The distribution of Scots pine crops in the four height strata considered is similar to that found in Scots pine mature stands in the British lowlands bioclimatic zone (Ferris-Kaan et al. 1998).

3.3 Coarse Woody Debris

The characterisation of CWD in the plots is presented in Table 4. The volume of CWD is higher in Valsaín than in Navafría, with more CWD types and rot classes. The low quantities

Table 4. Coarse woody debris (CWD) per types and rot classes in Valsaín and Navafría plots. Dead standing trees and stumps are quantified by number of items/ha and logs and branches by volume (m^3/ha).

CWD class	Size	NAVAFRÍA					VALSAÍN				
		Rot class									
		I	II	III	IV	V	I	II	III	IV	V
Dead standing trees (n/ha)	∅≥10 cm			0					4.0		
	10 cm≥∅>5 cm			0					0		
Stumps (n/ha)	∅≥30 cm	0	6.0	0	18.0	72.0	16.0	24.0	14.0	12.0	2.0
	30 cm>∅≥5 cm	0	0	0	0	0	0	25.0	37.5	62.5	25.0
Fallen logs and branches (m^3/ha)	∅≥10 cm	0	0	0	0.24	0.03	0.14	0.14	0.02	0	0
	10 cm>∅≥5 cm	0	0	0	0.10	0.04	0	0	0.13	0.03	0.01

of CWD in Navafría and the absence of some types of CWD, as stumps smaller than 30 cm, is explained by the intensive thinning program carried out in this forest.

4. Concluding remarks

These are the preliminary results of a permanent trial which will provide us information on the variation of structural diversity through the development phases. Each one of the analysed indices provide a valuable information about the forest structure. For this reason we think fit the join use of all of them to tackle these kind of studies.

We have identified in the 80–100 age class slight differences in some components of the structural diversity. This could point to the structural differences that the regeneration method produced initially decline with time elapsed, at least in this case, with large rotation period. It is necessary to analyse all plots in order to explore adequately the variation of the structure throughout the forest dynamic under the two silviculture systems.

In the future, the study of the relationships between flora, genetic and stand structure diversities we will enhance our knowledge of the effect of silviculture and regeneration method used on biodiversity. In the future, other components of biodiversity should be incorporated into the trial to improve the scientific basis for sustainable management of Scots pine forests in Spain.

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Monitoring of Forest Condition in Europe

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Abstract

Forest condition in Europe was started to be monitored 18 years ago jointly by the International Co-operative Programme on Assessment and Monitoring of Air Pollution Effects on Forests (ICP Forests) of the United Nations Economic Commission for Europe (UNECE) and by the European Union (EU). Crown condition shows a high spatial and temporal variation which is explainable mainly by tree age, weather extremes, biotic factors and air pollution. Defoliation of all trees observed continuously between 1989 and 2002 increased, however, with great differences among the individual species. Results of deposition measurements reflect that sulphur depositions decreased whereas nitrogen depositions remained nearly unchanged over the last decades. The average depositions of nitrogen exceeded those of sulphur, indicating that acidity related to ammonia has become a prominent source of acidification.

Keywords: forest condition; defoliation; weather; biotic factors; air pollution; nitrogen deposition

1. Introduction

Forest condition has been an issues of interest to forest scientists and practical foresters ever since. It received increasing attention in the early 1980s as a response to growing concern that defoliation in parts of the forests in Europe could be caused by air pollution (e.g. Ulrich 1981; Schütt 1982). Since then forest condition has been a subject of scientific, political and public debate, today being discussed within the wider context of sustainable forest management. However, there exists no generally accepted definition of the term “forest condition“ (United Nations 2000). It is often used synonymously with the terms “forest health” and “forest vitality”. In any case the term “forest condition” does not aim solely at the condition of forest trees, but at

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the condition of the forest ecosystem. The condition of a forest ecosystem is tried to be described by a set of parameters of its different compartments which are thought to permit an assessment of the ecosystem's long-term stability and the numerous factors influencing it. The most important factors are weather extremes (e.g. drought, heat, storm and frost), biotic agents (insects and fungi), game, fire, forest management and air pollution. These factors hamper sustainable forest management and hence the ecological, economic, social and cultural functions of forests for the human society. International environmental politics aim at preventive measures relying upon a sound scientific basis. A corner-stone of this scientific basis is the long-term monitoring of forest condition.

Forest condition in Europe was started to be monitored 18 years ago by the International Cooperative Programme on Assessment and Monitoring of Air Pollution Effects on Forests (ICP Forests) under the Convention on Long-range Transboundary Air Pollution (CLRTAP) under the United Nations Economic Commission for Europe (UNECE) in close cooperation with the European Union (EU). Today, 38 European countries as well as Canada and the United States of America are participating, rendering the monitoring programme one of the greatest of its kind worldwide. The 25 EU-Member States conduct the monitoring under the Regulation (EC) no. 2152/2003 "Forest Focus" of the European Commission (EC). EC supports ICP Forests financially and contributes greatly to the management and evaluation of the monitoring data. Canada and the United States of America do not use the same monitoring methods as the countries in Europe. However, they contribute national reports and are increasingly involved in the further development and joint application of monitoring methods in some fields of the programme. The main objectives of the programme are

- to provide knowledge of the spatial and temporal variation in forest condition on the European scale and their relationships to environmental factors;
- to contribute to a better understanding of the relationships between the condition of forest ecosystems and both natural and anthropogenic stress factors (in particular air pollution) throughout Europe using a network of sample plots.

The infrastructure, data base and results created by ICP Forests and EC permit increasingly contributions to forest policy at national, pan-European and global level on the effects of climate changes on forests, sustainable forest management and biodiversity in forests. The latter political processes will be covered in particular by EC under its Regulation "Forest Focus". ICP Forests is also contributing to these processes, but is primarily contributing to the work of CLRTAP of UNECE.

2. The monitoring system

The above mentioned two objectives are implemented by means of a systematic large-scale monitoring network and an intensive forest monitoring programme. The large-scale monitoring (Level I) aims to assess the spatial and temporal variation of forest condition across Europe. It is therefore pursued on a large number of monitoring plots. Given the large number of plots, only a limited set of parameters can be assessed and hence little evidence of cause-effect relationships can be expected at Level I. Cause-effect relationships are the target of the intensive monitoring programme (Level II) with its much larger set of monitoring parameters. The labour and cost intensive monitoring restricts the number of Level II plots. The numbers and the locations of the Level II plots were chosen by each country according to international guidelines and national priorities. The intensive monitoring aims at the ecosystem scale rather than at the European-wide scale.

At Level I approximately 6000 permanent plots are systematically arranged in a 16 x 16 km transnational grid. In a small number of countries the plots are arranged randomly instead of systematically, but the plot density corresponds to that of the 16 x 16 km grid. In parts of Scandinavia even the density of the plots is smaller which results in an under-representation of this part of Europe in the total Level I plot sample. On all Level I plots annual crown condition assessments are carried out. Also soil surveys were conducted on 5289 plots, most of them in the years 1993–1995. A repetition of the soil survey is planned for 2006. Moreover, foliage surveys were conducted on 1497 plots, most of them in the years 1992–1997.

For the intensive monitoring more than 860 Level II plots were selected in the most important forest ecosystems of the participating countries. The intensive monitoring aims at crown condition, soil condition, soil solution chemistry, foliage chemistry, tree growth, tree phenology, ground vegetation, meteorological condition, ambient air quality and deposition. Not all of the respective monitoring activities are conducted on all Level II plots (Table 1). The crown condition surveys on the Level II plots and on about half of the Level I plots include the assessment of several identifiable damage types such as insects, fungi, game, fire and abiotic agents. For Level II also the assessment of litterfall is foreseen and the respective method has been developed. All surveys within the programme are based on harmonised methods documented in a regularly updated manual (UNECE 2001). Since the establishment of the programme a comprehensive data bank on a wealth of monitoring parameters has been built up.

In each participating country the responsibility for the surveys lies with the national forest services. All countries are represented in the Task Force of ICP Forests which is chaired by Germany. For the coordination of parts of the monitoring, of the data management, of the evaluation and of the reporting, Germany hosts the Programme Coordinating Centre (PCC) at the Federal Research Centre for Forestry and Forest Products (BFH) in Hamburg, Germany. In addition, the EU-Member States are represented in the Standing Forest Committee (SFC) of EC. Within the close cooperation of ICP Forests and EC, both the Task Force and the SFC share the decision-making power for the total common monitoring programme. It is expected that EC will take over important parts of the data management and evaluation in the future. EU-Member States receive co-financing of the surveys from EC.

3. Crown condition

Crown condition is a fast reacting indicator for numerous environmental factors affecting tree vitality. It is assessed by means of visual assessments of defoliation and discolouration which is an inexpensive method permitting about 135 000 sample trees to be assessed annually on the approximately 6000 plots of the transnational grid. The drawback of this approach is that the assessment results are influenced by the subjectivity of different observers. Several data quality assurance measures were therefore introduced. At the national level, observer bias is estimated by analysing training and test results as well as results of control assessments (Schadauer 1991; Köhl 1991 and 1992). A high standard of training of the assessors can reduce observer bias. For individual species, the possibility to reach reliable results of the defoliation assessments at the national level has been shown (e.g. Eichhorn und Ackerbauer 1987; Dobbertin et al. 1997). At the international level the assessment results show systematic inconsistencies between different countries (e.g. Innes et al. 1993). Several efforts are undertaken to identify and reduce such systematic inconsistencies by means of cross-calibration and inter-comparison courses as well as by means of photographic techniques.

Those trees of the six most frequent species having been assessed continuously at Level I between 1989 and 2002 reveal in general increasing defoliation, however, with great

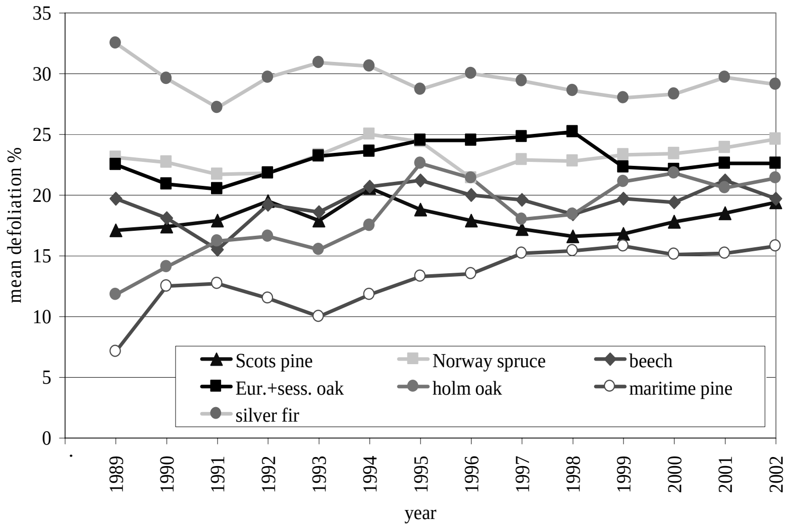

Figure 1. Development of mean defoliation of the 6 most frequent species. Number of trees: *Pinus sylvestris*: 2521; *Picea abies*: 2988; *Quercus robur* and *Q. petraea*: 1237; *Fagus sylvatica*: 2620; *Pinus pinaster*: 1360; *Quercus ilex* and *Q. rotundifolia*: 2243.

differences between individual species (Figure 1). The increase is very obvious for *Pinus pinaster* as well as for *Quercus ilex* and *Quercus rotundifolia*. Defoliation of *Pinus sylvestris* and *Piceas abies* was higher in 2002 than in 1989, however, showing large annual fluctuations over the period of observation. *Pinus sylvestris* recovered markedly from its high defoliation in 1994, however, its defoliation has been increasing again after 1998. No trend at all is revealed for the defoliation of *Fagus sylvatica* as well as for *Quercus robur* and *Quercus petraea*. The latter two species reveal an obvious recovery from a high of defoliation in 1998 (Lorenz et al. 2003).

The mean development of defoliation across Europe (Figure 1) reflects neither the high spatial variation of defoliation and its development nor any causes related to it. In recent multivariate and geostatistical studies (Lorenz et al. 2002), both the temporal and the spatial trends in mean plot defoliation of *Pinus sylvestris* (1313 plots) and *Fagus sylvatica* (399 plots) were evaluated in relation to

- the presence of biotic agents (insects and fungi) according to crown condition assessments;
- the amount of precipitation from January to June provided by the Global Precipitation Climatology Centre (GPCC);
- the deposition of S, NO_x and NH_y as modelled by the Cooperative Programme for Monitoring and Evaluation of the Long-range Transmission of Air Pollutants in Europe (EMEP).

The only significant – but weak – statistical relationship found is the positive correlation between defoliation of *Pinus sylvestris* and sulphur deposition. This is explained by the high number of *Pinus sylvestris* plots in areas of previously high defoliation and sulphur depositions particularly in parts of Poland, the Czech Republic, the Slovak Republic and the Baltic States. Figure 2 shows the decrease in defoliation of *Pinus sylvestris* in these areas from 1994 to 1999. Comparatively small areas of increasing defoliation in Bulgaria and

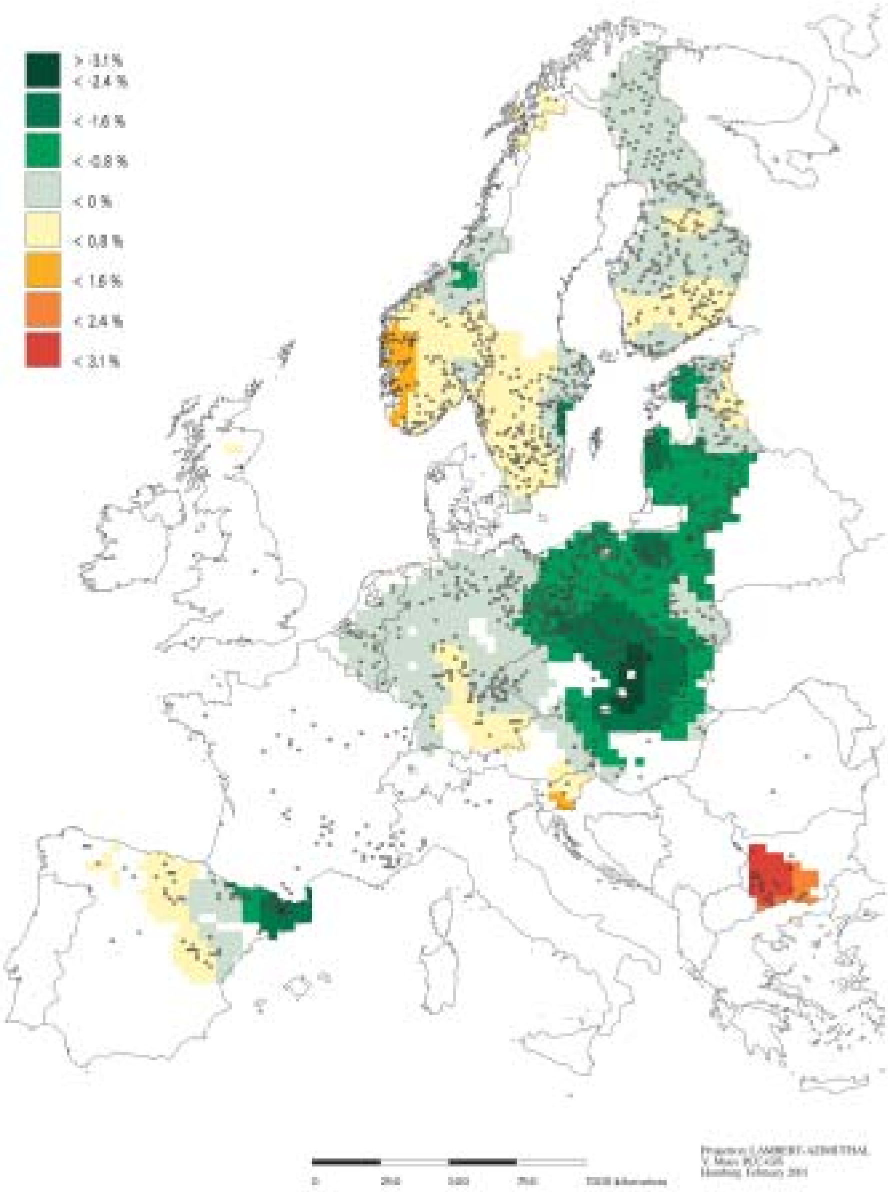

Figure 2. Kriged temporal trends of defoliation (changes in defoliation in percent points) of *Pinus sylvestris* from 1994-1999 (from Lorenz et al. 2002)

Norway were explained by weather extremes and *Gremmeniella abietina* attacks, respectively, by the national forest services. The temporal trends in the defoliation of *Fagus sylvatica* show a more heterogeneous spatial pattern with several spots of improvement and deterioration scattered across Europe. Frequently reported stressors influencing the condition of *Fagus sylvatica* are drought and frost. The spatial variation of defoliation of both *Pinus sylvestris* and *Fagus sylvatica* was largely explained by stand age and by the variable

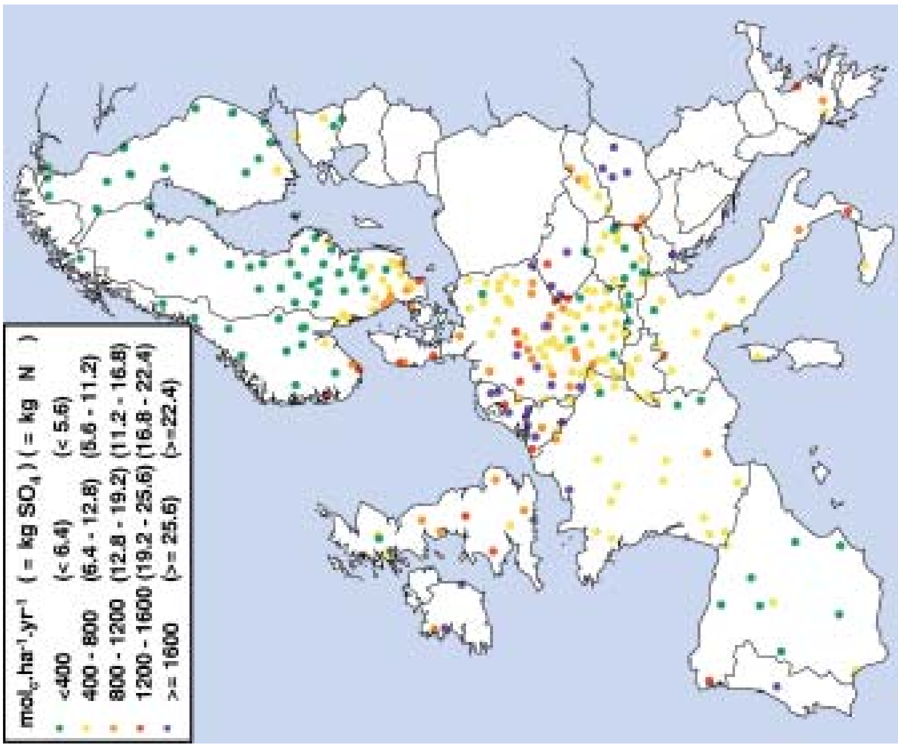

Figure 3. Average annual SO_4 deposition 1995-1998 (from De Vries et al. 2001).

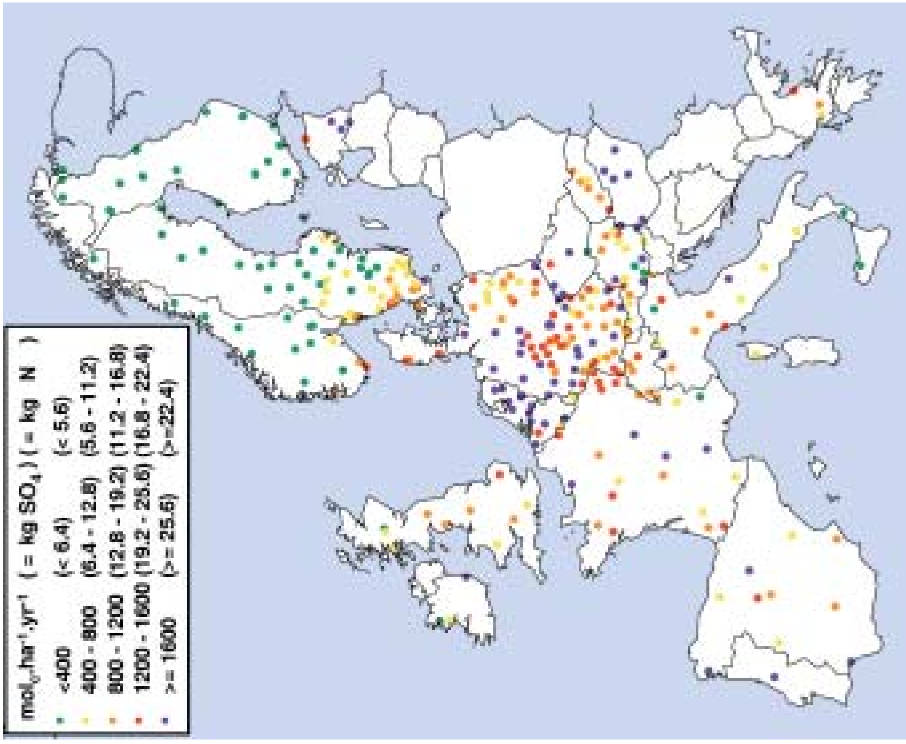

Figure 4. Average annual N deposition (NH_4 + NO_3) 1995-1998 (from De Vries et al. 2001)

"country". The correlation with the latter variable reflects partly the above mentioned systematic methodological differences between countries. The correlation between defoliation and stand age has been recognized since the first crown condition surveys. It seems plausible that this reflects at least partly the natural thinning of the crown with increasing tree age.

Since the early 1980s, defoliation observed in many forest areas of Europe has been discussed in connection with tree dieback and losses of forest growth. The monitoring of crown condition at Level I has not revealed an increase in tree mortality at the large scale (Lorenz and Becher 1994; Lorenz et al. 2000). Also the removals of trees were not increased and remained comparable to e.g. German thinning regimes. Resulting from large-scale monitoring, these findings do not reflect the local dieback observed in certain main damage areas.

As regards forest growth, defoliation has long been known to cause decreases in increment. This negative correlation between defoliation and increment was recently confirmed by two studies based upon Level II growth data for *Pinus sylvestris*, *Picea abies* and *Fagus sylvatica* (Fischer et al. 2004). Another study also involving Level II growth data found that the growth of younger trees of the same species was higher than that of older trees when they were of the same age (e.g. Kahle et al. 2004). Increasing growth over time and decreased growth due to defoliation are not necessarily contradictory to each other. While enhanced site productivity due to e.g. forest management practices, nitrogen deposition, higher temperatures and higher availability of CO_2 may increase the growth level of a stand, defoliation of certain trees can decrease the increment of these individuals and hence limit the stand growth.

4. Sulphur and nitrogen deposition

In the years from 1995 to 1998 deposition was measured on 309 Level II plots below the forest canopy (throughfall) as well as at nearby stations in the open field (bulk deposition). In addition, on parts of the plots the stemflow was assessed. Measurements of the stemflow were especially undertaken for *Fagus sylvatica* whose smooth bark markedly contributes to the total deposition. The total deposition of throughfall and stemflow was corrected for uptake and leaching of elements in the canopy by means of models. The substances assessed were sulphate (SO_4), nitrate (NO_3), ammonium (NH_4) and basic cations as the sum of calcium (Ca), magnesium (Mg) and potassium (K).

The evaluation of the deposition measurements by de Vries et al. (2001) confirms the pattern of high values for sulphate deposition in Central and Western Europe and lower ones in Scandinavia and south-western Europe. For 309 of the investigated plots the median for sulphate deposition (9 $kg \cdot ha^{-1} \cdot yr^{-1}$) is lower than the median for nitrogen deposition (14 $kg \cdot ha^{-1} \cdot yr^{-1}$) (Fig. 3 and 4, resp.), which leads to the conclusion that at present nitrogen is the most dominant source for the potential soil acidity.

5. Element budgets

Element budgets can be calculated for the forest ecosystem by subtracting the leaching of elements from the deposited amounts. The budget is positive if deposition exceeds leaching, i.e. the element is accumulated either in the soil or in the plants. In the case of nitrogen, however, the release into the air due to nitrification has to be taken into account. Vice versa, a negative budget indicates a release of a particular element from the ecosystem.

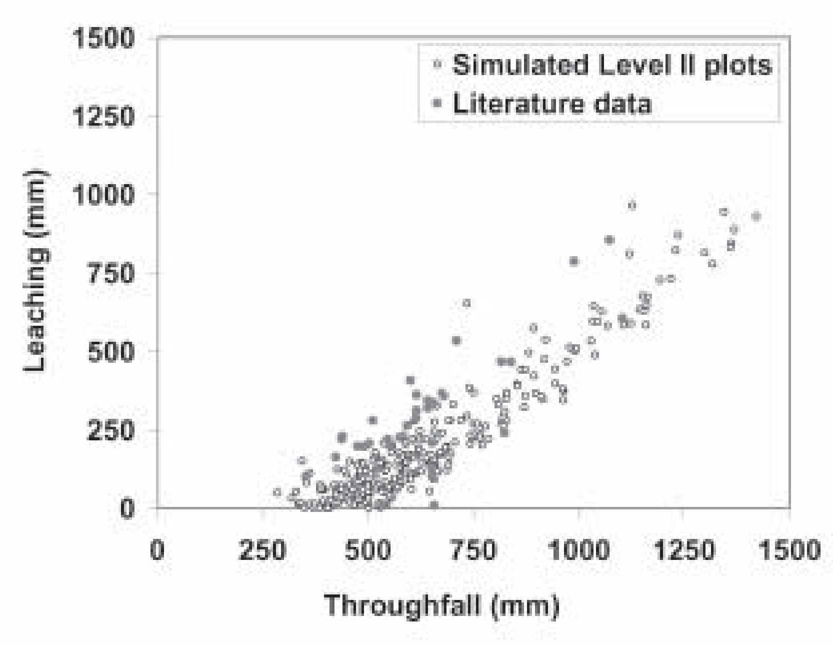

Figure 5. Simulated and measured annual water leaching fluxes (from de Vries et al. 2001).

The leaching of elements was calculated by de Vries et al. (2001) by multiplication of the measured element concentrations in the soil solution beneath the rooting zone with the amount of leached water. For this purpose, the amount of leached water had to be determined by use of a model. The model is based on the fact that the amount of leached water is a function of the precipitation in the stand and the evapotranspiration. Meteorological parameters such as precipitation, temperature, relative humidity and wind speed are taken into account. Figure 5 shows the modelled amounts of leached water versus throughfall for those 121 Level II plots for which the respective data were available. The modelled values are in good consistency with literature values which are also plotted for comparison.

Budgets were calculated by de Vries et al. (2001) for those 121 Level II plots for which deposition and leaching of sulphur and nitrogen were known. The spatial patterns of sulphur and nitrogen budgets are shown in Figures 6 and 7, respectively.

In total, the budgets of sulphate over all 121 plots are balanced. However, on 57% of the plots the budgets are negative. The plots with the highest release of sulphur (negative values) are located in central Europe where sulphate depositions were much higher some decades ago. It seems that sulphur being leached today was accumulated in previous episodes of high sulphate depositions. The decreasing sulphate deposition is reflected in decreasing sulphate concentrations in the bulk deposition assessed in the open field close to 285 Level II plots from 1996 to 2001 (Fischer et al. 2004).

In contrast, the leaching of nitrogen is generally much smaller than its deposition, indicating a current storage of nitrogen in the ecosystems. On 96% of the plots, nitrogen is either stored or released to the air. Ongoing monitoring and evaluations aim to clarify to which degree forest growth and the species diversity of ground vegetation are affected by eutrophication (Fischer et al. 2004).

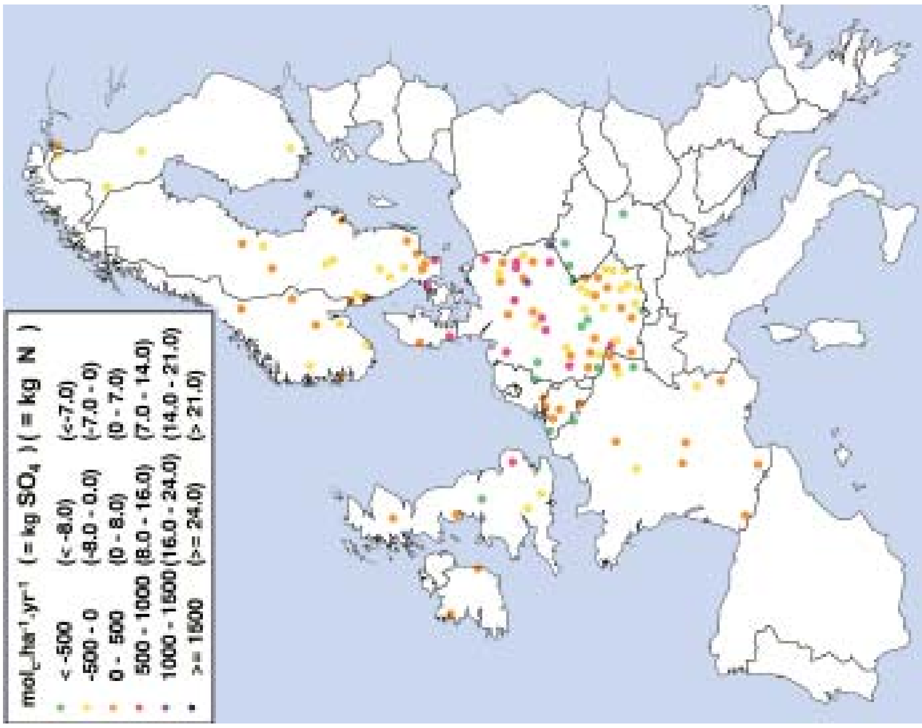

Figure 6. Annual average sulphur budgets 1995-1998 (from de Vries et al. 2001)

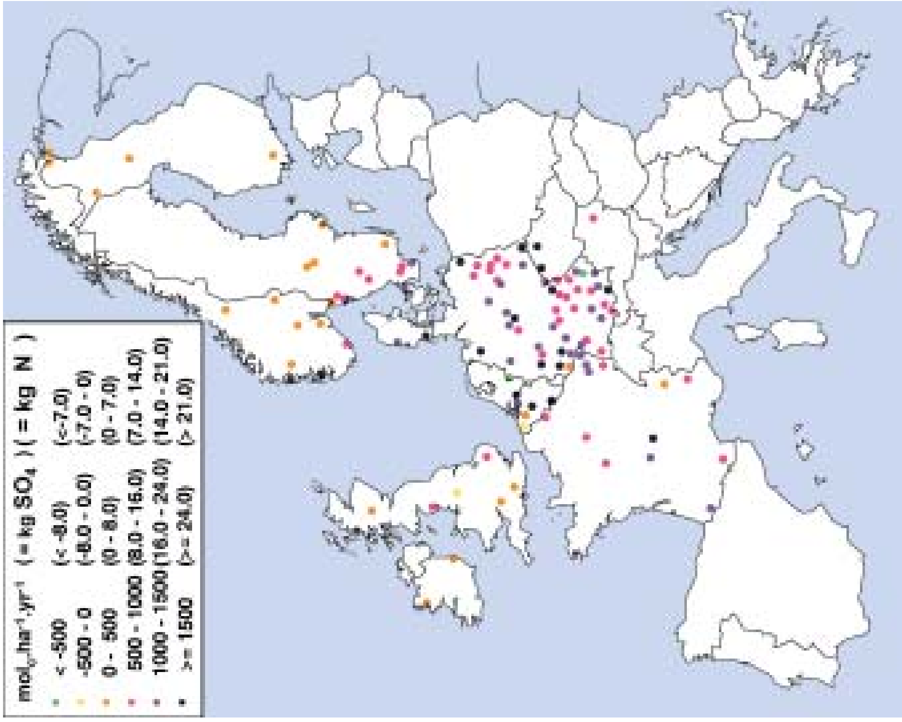

Figure 7. Annual average nitrogen budgets 1995-1998 (from de Vries et al. 2001)

6. Outlook

Forest monitoring in Europe will continue to provide the scientific basis for clean air policies under UNECE and EU. After first successes of clean air policies, the future tasks of the programme will comprise the verification of the effects of emission control. However, its well established infrastructure, its multidisciplinary monitoring approach and its comprehensive database permits significant contributions to other areas of environmental politics. Under CLRTAP, ICP Forests cooperates with related programmes (ICPs). Some of its intensive monitoring plots fall into catchments monitored by ICP Integrated Monitoring. Both programmes use the same methods of forest monitoring cooperate in the assessment of cause-effect relationships. In cooperation with ICP Modelling and Mapping, critical loads of depositions and their excesses are calculated for Level II plots (Fischer et al. 2002). The monitoring of ozone and ozone injury is conducted in cooperation with ICP Vegetation. (Fischer et al. 2003). In the field of global forest assessment, ICP Forests has contributed the results of the national crown condition assessments collected from its participating countries to the Boreal and Temperate Forest Resources Assessment of the Food and Agriculture Organisation (FAO) and UNECE of the year 2000 (United Nations 2000). The programme is pursuing the objectives of Strasbourg Resolution 1 (S1) of the Ministerial Conference on the Protection of Forests in Europe (MCPFE) and provides information on some of MCPFE's indicators for sustainable forest management. The programme's results on the diversity of forest plant species and the expected results from the planned monitoring of forest biodiversity involving besides ground vegetation also epiphytic lichens, stand structure and deadwood will provide information relevant for the implementation of the Convention on Biological Diversity (CBD) (Fischer et al. 2002). The soil and growth data permit estimations of the sequestration of carbon in forest soils and trees (Fischer et al. 2000). Future integrative evaluations of meteorological measurements, phenological observations, increment measurements and crown condition assessments may reveal the impact of climatic factors and contribute to the discussion of climate change. Such results will be of relevance for the Framework Convention on Climate Change (FCCC). In the field of air pollution effects on forests, ICP Forests has – besides its work under CLRTAP – also contributed to a report of the United Nations Forum on Forests (UNFF).

The programme is receiving increasing attention from policy-making bodies and research institutions outside Europe. This is demonstrated by the recently launched cooperation with North American forest monitoring programmes in the field of critical loads assessments. Another example is the discussion of the applicability of European forest monitoring approaches to East Asian forests with the Acid Deposition Monitoring Network in East Asia (EANET) (Fischer et al. 2004).

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Environmental Economics and Sociology

Forest Policy Developments in Changing Societies: Political Trends and Challenges to Research

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Abstract

The paper shows the increased complexity of political processes and forestry issues in international, European and national policy development. It discusses different dimensions of a systematic approach in developing socio-economic research on human environment system interactions addressing sustainable land management at ecosystem, landscape and ecosphere levels. It concludes that an interdisciplinary research approach combining natural sciences and socio-economic disciplines is essential in order to build a more permeable science-policy interface, to gain more knowledge about human-environment system interactions, and to provide tangible and useful information to politicians and the public.

Keywords: forest research; ecosystem research; human-environment system interactions; forest policy; sustainable land management.

1. Introduction

Sustainable development, balancing economic, social and environmental goals concerning renewable natural resources, is today the overarching principal of forestry. Changes in the attitudes of the public towards forest and new political actors pressing for more emphasis on the their social meaning as an integral part of landscape and environment are driving forces which demand new approaches in the protection, use and management of forest ecosystems, and more public participation in making decisions on their use and management. Within this context the paper addresses the following two questions:

- What are the significant political processes that have occurred over the last years and what are the present trends in public policy making with regard to improved forest protection, conservation and development?

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- What are the implications for forestry and ecosystem research in order to gain more knowledge on the complexity of human-environment system interactions and to make politically relevant recommendations for sustainable land management practices at landscape and ecosphere levels?

The first part of the paper focuses on the global need for maintaining a sustainable resource base as stipulated by the World Summit on Sustainable Development in 2002 which places forests and forestry development clearly into a multi-sector context. It emphasises the European scale of the forest sector and the importance of political initiatives and measures that occur at the European level. The second part provides a brief diagnosis of the dynamic changes and trends in national forest policy developments as they can be identified in many European countries and in particular in the countries of Central and Eastern Europe. The third part of the paper deals with the issue of how the increased complexity of forestry issues and political processes can be met by new interdisciplinary forest research focusing on human environment system interaction at different spatial scales and integrating more consistently natural sciences research with research on the cultural, social, economic and political dimensions of sustainable land management.

2. Forestry Development in a Global and European Perspective

The Global Context of Maintaining a Sustainable Resource Base: The World Summit on Sustainable Development, which took place in Johannesburg, South Africa in 2002, reconfirms the outcomes of the major United Nation conferences and international agreements since Rio. It places forests into a multi-sector context as an important part of the renewable sustainable natural resource base. It acknowledges the multiple and varying outputs from forests for poverty alleviation, as raw material and energy resources, and as natural habitats and environment. Achievements of sustainable forest management, nationally and globally, through partnerships among interested governments and stakeholders, are essential goals of sustainable development. This includes the private sector, indigenous and local communities and non-governmental organisations.

The Plan of Implementation puts strong emphasis on an integrative approach in protecting and managing the natural resource base as a whole and states that human activities have an increasing impact on the integrity of the ecosystem (WSSD 2002). It underlines the necessity to implement strategies, which are based on targets adopted at the national and/or regional levels in order to protect ecosystems and to achieve an integrated management of land, water and living resources. The Plan highlights the role of forests in important policy domains such as natural resources management (Section 23), agriculture (Section 38d), desertification (Section 39d), and mountains (Section 40b). It also shows that land use and forest management decisions have substantial links to political decision addressing measures on climate change (Section 36), maintaining biodiversity (Section 42), and the institutional framework for sustainable development (Section 120 and following).

The establishment of a constitutional framework and of a public security system provides the foundation for state interventions through guaranteeing the rule of law (von Prittwitz et al. 1994). Both are fundamental to legislation regulating specific policy domains. Laws on economic production and technology development have strong forward linkages to the sector and cross-sector laws inasmuch as the natural resource base and environment have an important impact on economic activities. Laws promoting development, security of subsistence and well-being of people depend to a large extent on backward linkages to social and economic policies as well as on the constitutional framework that regulates, for instance, ownership rights and entrepreneurial activities. Sector laws and policy programmes show a high degree of positive and negative connections among each other.

Figure 1 shows different kinds of public policies and laws at stake, depending on the particular situation of a country, in order to achieve an integrative approach in protecting and managing the natural resource base.

- Policies and laws establishing a constitutional framework and a public security system guarantee the rule of law, provide a foundation for private activities and entrepreneurship, and are fundamental to define state competencies and the content of public policy domains.
- Economic, trade and finance policies and legislation that establish a framework for socio-economic production and cultural integration have strong backward links to the constitution and important forward links to sector and cross-sector policy programmes.
- Laws and policies promoting development and security for subsistence, for instance, through technological innovation, research and education, and through environmental protection have important feed-backs to economic productivity, income generation and social integration.

The International Forest Regime: An expression of the global political context in which forests are now placed is the emerging international forest regime which is based on five main pillars:

- International legal instruments such as conventions, agreements and declarations addressing forests and forestry directly or indirectly;
- World-wide political processes within the United Nations System involving
- governments, non-governmental organisations, the private sector and indigenous and local communities;

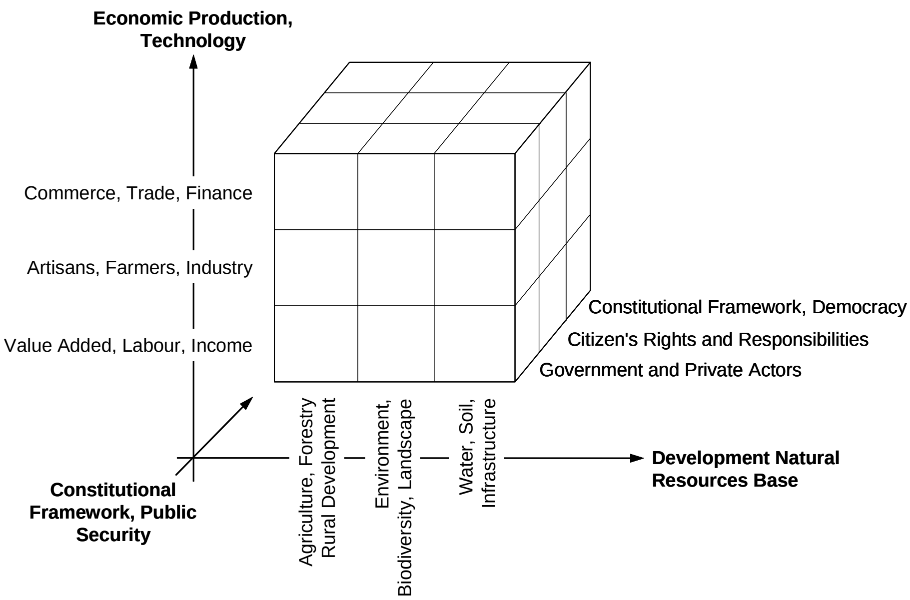

Figure 1. Different categories of laws in maintaining the natural resource base.

- Regional forest-related initiatives which operate at present in several continents and develop their own political agenda on forests and the forest sector;
- International Criteria and Indicator Processes which provide eco-region specific standards for sustainable forest management certification systems;
- National Forest Program Processes which are largely based on stakeholder concertation and allow to concretize international commitments of individual countries.

A substantial expansion of international laws on environment and development has taken place, which establishes a multi sector framework for forestry (FAO 2002). The Forest Principles are innovative and comprehensive by intention but non-binding. The principles contribute to make international discussions on forests more substantial and have probably changed the thinking of many professionals. However, there remain at present significant gaps between the non-binding statements on forest protection and management and the goals and formal obligations of the three conventions adopted at the Rio Conference in 1992 which are the frame convention on climate change followed by the Kyoto protocol, the convention on biological diversity, and the convention to combat desertification. This makes it difficult to transmit global and cross-sector objectives on climate change, biodiversity and desertification into consistent national policies of environmental protection and to support integrated approaches in promoting sustainable land management practices at local levels.

European Dimension of the Forest and Wood Products Sector: Europe's move towards progressive economic development, open civil societies, democratic rule and common political institutions has many faces. There is the Council of Europe in Strasbourg, which offers a political platform for more than 40 European countries. There is the European Human Rights Convention, which provides a common framework for fundamental rights of citizens. And there are many trans-national and pan-European institutions and processes that deal with economic, social and environmental issues of common concern.

The steps towards more economic, social and political integration in Europe have far reaching implications for the forest and wood industry sector. For the wood products industry a continental European space offers opportunities and challenges such as (European Commission 2000):

- New and larger markets combined with structural changes in wood industries;
- More market and price competition inside Europe;
- A gain in efficiency and productivity in larger industrial units;
- Stronger positions but also more competition in world markets.

The impacts of the expanding network of political declarations, agreements and binding legal instruments that govern sustainable forest uses and management are manifold, challenging, and bring about new perspectives of European forestry development. They lead to:

- A new vision of the large variety of European forests;
- A more concrete understanding of common responsibilities;
- Forest ecosystem networks covering large European regions;
- Progressive adaptations of national policies and laws;
- Common management principles and standards;
- Increasingly integrated research and education networks at a European scale;
- A new role of European forestry and science in world forestry.

A Political Platform for European Forests: The most important Pan-European institution in forestry matters is at present the *Ministerial Conference for the Protection of Forests in Europe* which involves more than 40 countries, including the Russian Federation. It started with its first conference in Strasbourg in 1990 as a reaction to the years of forest threats due to atmospheric pollution. It developed rapidly to a common intergovernmental forum addressing fundamental economic, social and environmental issues. An important step was made at the 1994 Helsinki conference in developing a common definition of sustainable forest management. As a follow-up, an agreement was built on 6 relevant indicators combined with quantitative and qualitative criteria for evaluation, which were endorsed by a resolution of the Lisbon conference in 1998. The Lisbon conference adopted a second resolution focusing on human resource development and socio-economic issues. Sustainable management, national forest programmes and socio-economic concerns remain on the agenda (MCPFE 2003a).

The Vienna Conference of April 2003 followed this line under the far-reaching title "Living Forest Summit: Common Benefits, Common responsibilities". The general declaration and resolutions of the Vienna Conference show the actual common political dimensions of European forests and forestry and are a valuable information source for policy analysis (MCPFE 2003b). The five resolutions adopted during the Conference address cross-sector cooperation and national forest programmes, economic viability of sustainable forest management, social and cultural dimensions of forestry, forest biological diversity, and linkages between climate change and sustainable forest management.

The Role of the European Union: A corner stone contributing to cooperation in many domains and favouring a new European identity is the *European Union* itself. The Union is by no means Europe, neither in its extension, nor in its global richness, nor in its great diversity. However, its increasing momentum is a driving factor towards a more permeable and integrative continent. It is a continent in which people can move according to their personal choice and in which trans-national and national political institutions coexist. The European Union (EU) offers an interesting example where a supranational policy framework has gained considerable momentum and importance both for policymaking in the member countries as well as in international policy processes. With regard to forest conservation and forestry development, the EU example is of particular interest as its policy and legal framework relies increasingly on cross-sector measures (Cirelli and Schmithüsen 2000).

The supranational framework operates through Community Council regulations and directives, which are implemented by the member states either as direct EU regulations or by adjustment of national policies and regulations. Not having specific competencies in forestry matters, the EU has adopted numerous measures in other policy domains that have immediate and largely positive impacts on forests and forest management. This is particularly the case for policies relating to agriculture, rural development, nature conservation and environmental protection. Community programmes on technology development, consumer safety, research and development, and education are other domains of importance to the forestry and wood-processing sector.

Europe's Role in World Forestry: A new understanding of the European role and opportunities in the forest sector accrues from the continental scale of the forest resources. According to the most recent FAO statistics the total forest area in Europe is over a billion hectares or slightly more than a quarter of the world's forests (FAO 2001). Western and Central European forests extend over an area of 170 million hectares, of which 115 million are situated in the European Union (UN-ECE/FAO 2000). More than 80% of the total forest area belongs to countries of the Community of Independent States (CIS). And again the large majority of the CIS country forests (90%) are situated in Russia.

At the same time a new understanding of Europe's role in international forestry develops. In fact European engagements and initiatives make already important contributions to worldwide efforts to preserve rare and significant forest ecosystems and to manage production forests in a sustainable manner. The increasing political cooperation, the common concerns on solving environmental problems jointly, and the building of common research and teaching networks are important assets for the future. The European Union and the member countries concert their objectives and measures increasingly and are strongly involved in all important international processes and programmes of the UN System addressing environment and development.

3. Progressive Change in National Forest Policies and Legislation

Role of National Policies and Legislation: The fundamental issues of the meaning and significance of forests in a particular society and at a given time are in a process of continuous change. Problems that seemed solved in the past need now solutions that take the new economic, political and social context into account. Adaptation and innovation of forest policies remain on the political agenda in the European countries. Global and regional trends press for a continuous innovation in policy development. They determine the conditions of national policy making to a considerable extent; influence the attitudes and behaviour of citizens, land users and land managers; produce varying networks of political actors; and establish complex multi-sector and multilevel policy networks.

Forest Policies and Laws Addressing the Existence Value of Forests: In adapting to the principles of sustainable development, modern forest policies and legislation need to address the full economic, social and environmental value of forests. This implies a combination of resources protection, land use and land management rules (Figure 2):

- Protection regulations refer to maintaining environment and biodiversity, to nature and landscape protection, and to the preservation of cultural and spiritual values associated with trees and forests.
- Land-use regulations provide for zoning of forest land, control of forest clearing, protection of a permanent forest estate, and for the establishment of new forest resources through afforestation.
- Utilisation and management regulations determine responsibilities of forest owners with regard to sustainable production of wood and non-wood products, the protection of soil and water resources as well as public access to forests and recreational uses.

Reformulation of Forest Policy and Laws in European Countries: The public framework for protecting and managing forest resources as well as the corresponding laws has been revised in practically all European countries (Schmithüsen et al. 2000). Major changes are occurring at present in Central and Eastern European countries. In transition to an open civil society, democratic institutions and a market economy, they have a difficult task. They have to develop a completely new policy and legal framework for addressing agriculture and forestry, nature conservation and environmental protection (Mekouar and Castelein 2002). Changes to improve the resources utilisation framework in the European countries of the forest sector have been greatly influenced by the growing political and social concerns related to the prevailing forestry practices. Societal demands on private and public forests, together with responses from within the forestry community and from the public at large, have received considerable attention from politicians and the forest administration. Altogether forest policies in most European countries follow increasingly internationally agreed objectives.

Expanding and more Comprehensive Policy Objectives: The goals of forest policy have become more diversified and comprehensive. Moving from a perspective, which focused on wood as a sustainable resource, they now address a wide range of private and public goods and values and acknowledge the equal importance of production and conservation. Policy goals are incremental and refer to the role of forests as multifunctional resources; for their

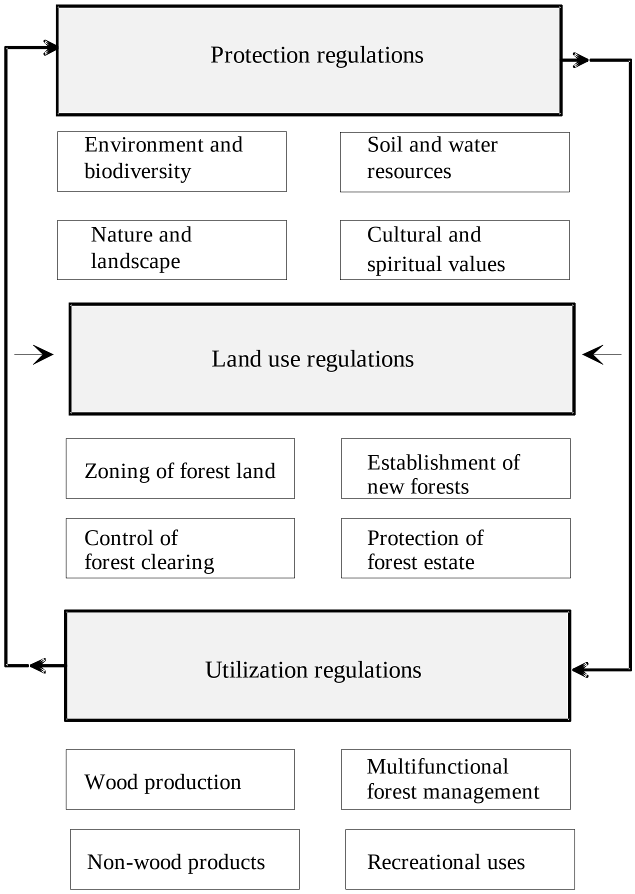

Figure 2. Forest policies and laws regulating protection, land use and utilisation.

economic potential and their importance for the environment. Increasingly they address the variety of ecosystems, the need to maintain biodiversity and the development potential of forestry in rural and urban areas. Similar dynamic processes with regard to the incremental objectives of policy and law have occurred since the 1970s in the USA and Canada (Schmithüsen and Siegel 1997).

Multifunctional and Close to Nature Forestry Practices: New and amended policies and laws favour multifunctional forest management as a land-use strategy that is capable of functioning among divergent social interests and local conditions. Multifunctional forest management practices are highly consistent with the principle of sustainable development and imply foremost:

- Decision-making processes involving forest owners, the principal users and environmental groups on an equal footing;
- New balances between private and public interests and the elaboration of workable arrangements for landowners facing public demands;
- A shift from governmental and hierarchical regulatory systems to negotiation, public process steering and joint management responsibilities;

- Realistic financial arrangements involving market proceeds, public funding and contributions from private user and interest groups to provide multiple forestry outputs.

Close-to-nature forestry practices are another land management strategy that contributes to maintain biodiversity, variety of ecosystems and diversified landscapes. It favours flexible and long-term production cycles, offers attractive areas for recreation and leisure activities, and leaves options for future uses and developments. In relying on natural site factors, close to nature forestry combines more consistently than other management practices economic necessities with multiple social and environmental requirements.

Joint Private and Public Management Responsibilities: The incremental role of public policies addressing forest protection and management makes it necessary to redefine the roles of the private and public sectors; to find equitable and effective balances between the benefits for, and responsibilities of stakeholders; and to adapt the role of government from intervention to process steering. New forms of joint management responsibilities for the forests in rural and urban areas need to be agreed upon in order to maintain economic benefits from wood production and processing, to safeguard the environment, to protect flora and fauna, and to preserve the cultural heritage, which forests represent in our societies. Cross-sector policy linkages and multi-sector policy networks are of fundamental importance in order to manage forest ecosystems and landscapes in a sustainable manner (FAO 2003; Schmithüsen 2003).

Multilevel Policy Networks: The combination of global, European and national commitments leads to an increasingly complex framework of policies with multilevel impacts. At the global level, free trade, environmental protection and biodiversity are dominant subjects. Forest-related aspects are increased industrial uses through access to new areas, reduction of large-scale deforestation, and conservation of natural forests. At the supranational level, major issues are structural changes in agriculture, and the protection of environment and water resources. Afforestation of marginal land and criteria and indicators for sustainable forest development are of importance. At the national level, an emphasis is laid on forestry and wood processing as productive sectors of the economy, and on the regulation of forest management practices. At the local level, multiple forest uses providing employment, protection and recreation are of immediate concern.

4. Challenges to Research on Forest and Landscape Management in Changing Societies

The diagnosis of new trends in forest policy and law developments shows a growing complexity of forestry issues and political processes relevant to forests. The changing conditions for sustainable forest management are to be seen in the overall perspective of maintaining the natural resource base, in a holistic understanding of forests and landscapes, and as part of the overall goal to protect environment and improve quality of life for present and future generations. This is in fact the central theme of wise use of forests and ecosystem management that builds on the legacy of the past and provides opportunities for the future (Farrel et al. 2000).

Research addressing relevant issues of forest policy development is today part of interdisciplinary scientific collaboration on ecosystem and landscape management. Significant policy research issues are:

- What are the driving factors for policy adaptation and change, and to what extent do present policies and legislation take up the challenge of change and innovation?

- What are the implications of policies and laws on multifunctional and sustainable land-use practices and what are the ways and means to foster such practices?
- Which positive and negative linkages exist between forestry policies and other public policies addressing environmental protection and natural resources utilization and what are the ways and means to built comprehensive and consistent policy networks?
- To what extent do policy regulations address the value of forests in a comprehensive manner and allow for reasonable balance between public and private interests?
- What are the conditions for building consensus among stakeholders involved in land management and what are appropriate instruments to foster participation and to develop agreed solutions?
- What are appropriate political and institutional requirements and what policy instruments can be selected in order to promote and support locally adapted land management solutions?

As for other land management sectors, sustainable development is today the overarching political principle and the benchmark for judging to what extent the forest sector and forest policies contribute to economic and social welfare and to a safe environment that benefit present and future generations. The essential content of this principle is that economic growth, social integration and caring for a liveable environment are on an equal footing. Economic growth, social integration and protection of the environment depend on each other, cannot be substituted for, and are fundamental to social progress and common welfare. The principal of sustainable development and the more specific political commitments of countries and the international community set the task for research and science.

The development of an integrative perspective to gain more knowledge about the interactions among social systems and human behaviour, ecosystem processes and environmental change is essential in order to understand more closely the impacts and feedbacks between man and his natural resource base. With regard to forests one has to understand the interactions between society and forest ecosystems, their social and cultural meaning, their potential for providing different combinations of goods and services, and their stability and biodiversity under alternative management systems (Piussi and Farrel 2000; Führer 2000).

Figure 3 indicates different dimensions of a systematic approach in order to develop socio-economic research on human environment system interactions at various scales. It relates dynamics and change as constituting elements in all societies to impacts on and feedbacks from the renewable natural resource base. It considers as a third dimension objectives, technologies, instruments, outcomes and feedbacks as key elements in public policy development and private and collective decision making processes.

The *first dimension* deals with change in societies. Cultural values expressed, for instance, in personal life styles and spiritual convictions, and social demands that relate to individual freedom, democratic participation and political organization are important driving factors that induce and reflect dimensions of societal change. In combination with changing economic needs and opportunities to produce multiple goods and services they initiate continuously changes and innovations in the prevailing political and legal systems. And altogether, these factors determine to a large extent individual and collective decision-making processes in natural resources utilization and management with landowners and land users as important primary agents.

The *second dimension* addresses present and likely future reciprocal interactions between human interventions and the renewable natural resources base. This includes global and regional environmental interactions; interactions at the level of landscapes, ecosystems or watersheds; interactions that result from alternative or combined land use systems; and interactions at the level of individual or corporative ownership and land management units.

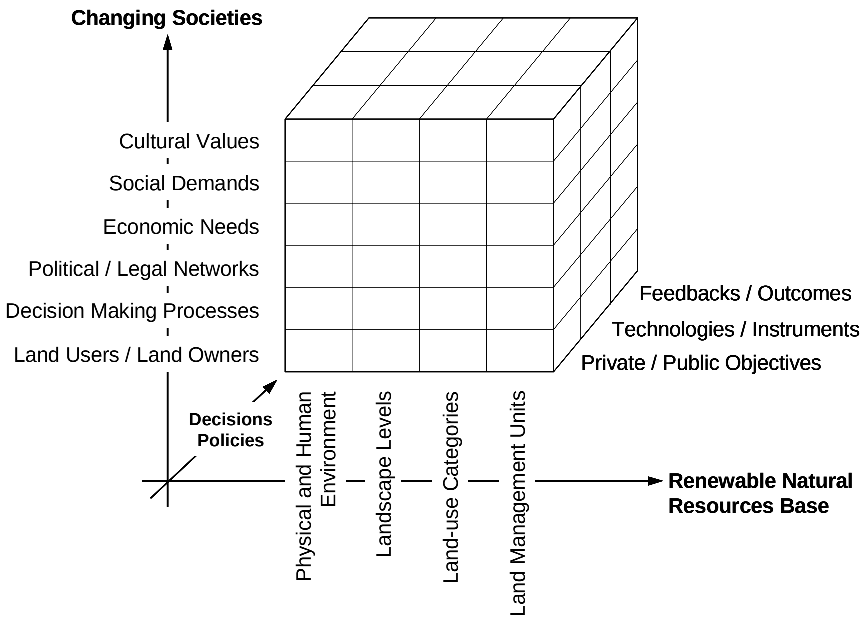

Figure 3. Human environment system interactions related to land management at ecosystem, landscape and ecosphere levels.

Significant elements of the *third dimension* are, for instance, the relationship between private and public interests and objectives; existing and new technologies in land management; and available tools such as effective and cost efficient political and economic instruments, and the feedbacks and outcomes from individual decisions and public policy measures. The latter are important signals to land managers, stakeholders and policy makers which show whether the taken course of action leads to satisfying results or requires corrections and further intervention.

The options which individuals and societies have and the choices they make in land uses and land management practices depend on complex interactions between demographic, socio-economic, political and institutional, physical and biogeochemical, and biological factors. One of the significant challenges to the research agenda is the need to explain the reciprocal links between environmental changes and different land use systems and to analyse the implications for appropriate choices and land management decisions in ecosystem and landscape management.

Multidisciplinary research is essential in order to identify the factors of change within the reciprocal relationships and to evaluate in quantitative and qualitative terms the effects and dynamics of human-environment system interactions. On the basis of such an analysis substantive proposals for improvements in land management practices can be made which are commensurate with the demands of landowners, land users and society as a whole. This implies to investigate the characteristics, dynamics and vulnerability of human-environment systems in a global and regional context, as well at the level of landscapes, distinct land-use categories, ownership and land management units.

A comprehensive understanding of human environment system interactions implies to acknowledge the cultural, social, economic and political dimensions of society respectively the societal norms which they induce as permanent factors of change. This demonstrates the obvious need for an interdisciplinary research approach combining social and cultural sciences, political and economic sciences, and decision making sciences. Such research has its own disciplinary methodological basis and needs to facilitate the integration of quantitative and qualitative knowledge (Scholz and Tietje 2002):

- *Socio-empirical and cultural research* is required, for example, in order to gain an understanding of the attitudes, perceptions and levels of acceptance of individuals, stakeholders, and societal groups; to show the variety of personal and collective values and their dynamics of change, or to identify motives and objectives in making concrete decisions in land management.
- *Policy and decision making research* refers, for instance, to participatory mechanisms and stakeholder involvement; to different forms of governance and political organisation; distribution of competences at different levels of government; decentralisation and political process steering; and to cross-sector policy effects and appropriate forms of political coordination.
- *Inputs from economic disciplines* and in particular from environmental and natural resource economics are necessary in order to investigate positive and negative external effects in quantitative and monetary terms. It is essential to determine ways and means for an effective internalisation of positive and negative effects, to identify trade offs between different categories of land use and forest conversion, to assess cost-benefits and cost-effectiveness of alternative land use management systems, and to quantify transaction costs that result from different land tenure systems.

The three dimensions indicated in Figure 3 present a framework to structure important human-environment system interactions at the ecosphere and landscape levels. They allow to identify a critical path of socio-economic conditions, regulating mechanisms and tools of intervention which are crucial in order maintain or to foster sustainable utilization of the renewable natural resources base. This again permits to construct analytical action oriented frameworks that show critical positive and negative feedbacks between human and physical regulation systems. A selective approach focusing on key points in public and private decision-making processes increases the practical relevance of the frameworks and models. The reciprocal relationships may be analyzed at the scale of ownership and management units, at different landscape scales, or with regard to their regional or global importance.

It is a political necessity to built a more permeable science-policy interface and to develop innovate and more comprehensive interdisciplinary research networks that are able to determine what impacts human interventions have on the environment, to assess individual and collective opportunities and risks that result from changes in the environment, and to show at the same time what concrete measures can be taken to increase benefits and to avoid or reduce risks.

5. Conclusions

Balancing economic, social and environmental goals in order to maintain and develop the natural resource base is now the overarching international and European requirement for forest protection and forestry development. At national level forest policies and law have incremental objectives and consider sustainable wood production, infrastructure protection, recreational use, nature and landscape protection and spiritual and aesthetic values in a more comprehensive perspective. Cross-sector policy networks and multi-level policy decision making processes determine increasingly the use and protection of forest.

The implications are far reaching and concern the role of forests, the goals of forest management, and the objectives of public policies addressing sustainable forest management.

- Public perception of the meaning of forests moves from a tradition sectoral view toward a global view of forests as economic resources, social space and a humane environment. Sustainable forest mangement is largely determined by local circumstances.
- Current forestry practices have to demonstrate that they are in accordance with a large range of public demands and values. They have to balance economic, social and environmental requirements as well as multiple and often divergent public and private interests.
- Forest policies are not anymore the exclusive public policy domain which addresses forest utilization and management. They can only be effective if conceived, formulated and implemented in the context of a growing number of public policies addressing rural development, nature and landscape conservation and environmental protection.

Forestry research today is a significant part of environmental, land management, and ecosystem management sciences. It can make a substantial contribution to the improvement of forest management practices if it considers human-environment system interactions as they affect forest and landscape management in changing societies. An interdisciplinary research approach combining socio-economic analysis and modelling of processes and interactions of the physical resource base can generate consistent advice and recommendations in order to improve land use planning and land management practices, and adopt more effective public policy measures and instruments.

The identification of critical factors determining human-environment interactions and choosing a critical path facilitates interdisciplinary research considerably. It allows to produce consistent, empirical and politically relevant socio-economic frameworks, which indicate critical factors that are of importance in a given context. The construction of frameworks showing the socio-economic context of sustainable resources utilization and land management is an important step to provide inputs into modeling and system analysis of physical processes and interactions of environmental change. It allows to conceive and design innovative research, which integrates systematically human and physical aspects in common models and system analysis and to build bridges between the natural sciences community, the environmental sciences community and the social sciences community. Combining modelling of physical effects with social, economic and political investigations on different kinds of land use and management practices creates added scientific value.

The question of how to produce such value leads to the more fundamental issue of how to establish appropriate working relations and institutional structures that foster creative and useful interactions between different classes of sciences. This requires, for instance, a continuous dialogue between scientists as well as research designs involving scientists and stakeholders interested in and concerned about forthcoming results. It needs to focus on concrete problems and field studies that are of political relevance in order to built an active research base. And it needs foremost to learn and to understand the science language of different disciplines, and the value and significance of the methods which they have developed and use. All this takes time and is difficult to accomplish. But it is essential in order to come forward with meaningful and practical results that are commensurate with the complexity of changing environmental conditions, productive land management, and wise and sustainable use of forests.

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Environmental Economics for Sustainable Forest Management

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Abstract

Environmental economics is not different from economics or forest economics but considers more thoroughly all the environmental and also social impacts of the decisions at hand. Environmental economics is concerned with all the sectors of the economy and as such is not much different from the new and broader concept underlying forestry economics as it has evolved in Europe.

This paper proposes a general framework for co-ordinating and organizing environmental and forest economics research activities in the form of a network of 'environment economists for sustainable development'. Its main objective would be to organize information, at both the micro and macroeconomic levels, needed for forestry resources decisions. The motivation behind this effort is to help showing where and how a particular research effort increases knowledge that overall could improve the decision- and policy-making processes. It could help also in coordinating research in forestry economics for sustainable management in Europe and elsewhere.

Keywords: Environmental and forestry economics; sustainable economic development; political economy of forestry; European research network.

Introduction

Recent re-awareness[1] of the fragility of economic growth in regards to the shrinking natural and social capital has been widely exposed for instance in Natural Capitalism (Hawken et al.

1 Awareness of environmental sustainability is not recent if we consider the work of classical political economists like Malthus and Ricardo or more recently with the Club of Rome, but the issue of sustainable economic growth has become of practical relevance at the global and political level more recently.

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EFI Proceedings No. 49, 2004

1999). The 3.8-billion-year store of natural capital is being exhausted faster than at any time in history. This is not an existence value problem but one of vital life giving services such as the water we drink and the air we breathe. Natural Capitalism recognizes the link between human-made and natural capital and identifies four types of capital: human, financial, manufactured and natural capital. An economic system uses the three first forms of capital to transform natural capital – here the forest is our main concern – into useful products such as paper, houses, furniture, fuel, medicine, and foods. But the climate debate is also a public issue in which the assets at risk are not specific resources like timber but a life supporting system. One of nature's most critical cycles is the continual exchange of carbon dioxide and oxygen among plants and animals. This recycling service is provided free of charge. By burning fossil fuel in the atmosphere, the capacity of forests and other ecosystems to recycle CO_2 is exceeded and there is no known alternative to nature's carbon cycle service.

It is in this broad state of mind that we want to propose the potential contribution of research in environmental economics to sustainable forest management. After reviewing the main idea behind sustainable economic development and its link to sustainable forest management, a tentative framework is suggested for a network of excellence in "Environmental Economics for Sustainable Forest Management" to organize European research effort in environmental/forestry economics. We conclude the paper with some brief remarks on possible priorities in environmental economics research for forestry in light of the Ministerial Conference on the Protection of Forests in Europe (MCPFE) proposals for Action.

Sustainable Economic Development

To define sustainable development is not easy since it has many dimensions that will evolve over time with technology, change in taste and aspirations, and the known resource base. Thresholds of forest resources utilization rates will vary with these parameters that are not constant but important for decision makers to know. A cursory review of the literature on sustainable economic development starting with the Bruntland Commission shows that the concept is not entirely new to foresters. However, this new political agenda has forced the use of new and more sophisticated tools to analyze forest decisions.

Definitions

The most frequently quoted definition of sustainable development has been given by the Bruntdland Commission (1987): "progress that meets the needs of the present without compromising the ability of future generations to meet their own needs." How to define needs is a question of political economy, way of life and culture. Environmental/Forest economists could talk about the needs to maintain the flow of consumption of all forest products and forest services and functions over time. Later definitions have retained the core ethic of intergenerational equity (Pezzey 1989) which puts to the fore the much debated discount rate in the forestry and environmental economics literature (Harou 1984). Good suggestions for research on sustainable development were provided by Brookfield (1990).

Recent definitions of sustainable economic development have focused more explicitly on the three pillars of sustainability: environmental, economic and social. While foresters have learned quickly to assess environmental impacts of their investments and policies, the economic and social impacts are still rarely assessed. However, the trend is reversing. It is the forestry/environmental economics research community that should help to put these techniques at the reach of analysts and decision-makers more widely. To maintain the flow of consumption for future generation depends on the change in stock of assets or capital: natural (environmental), man-made (economic) and human (social). Does the composition of this capital matter? In the environmental economics literature (Daly 1999; Ekins 2003), a distinction is made between weak sustainability in which these forms of capital are fully substitutable and strong sustainability in which these forms of capital are not substitutable. The latter is sometimes called ecological economics as opposed to neo-classical economics applied to the environment. For foresters this was translated in sustained yield even-flow and non-declining even flow policies (Luckert 2001). Nowadays in 'forest yield' is included also the forest flow of services and functions and we talk about maintaining landscapes, including different ecosystems and ecotypes, rather than forests. With the rapid demographic trends, economists had to recognize that technology could still increase the substitutability among different forms of capital but to only a certain point.

Substitutes and Irreversibility

For many essential environmental services, especially global life support systems, such as forest ecosystems, there are no known alternatives now and probably in the future as illustrated by the Biosphere 2 experiment (Heal 2000). The limits of substitutability among capital are probably greater for natural capital entering consumption untransformed: a natural forest scenery is not substitutable by other scenery. It is easier to substitute a produced output such as a wood window for example. Some natural capital has to be maintained especially if the drawdown entails irreversible loss and if it matters directly for the well-being of future generations. This concept includes the notion of the environment as a sink also, such as forest ecosystems as tertiary water pollution treatment (Stenger 2000). Given the uncertainty surrounding complex forestry ecosystems, environmental economists invoke the precautionary principle and stress how keeping option value of forest ecosystems for instance could make a serious difference in the future. Related to that, the notion of Safe Minimum Standard is useful (Criacy-Wantrup 1968; Harou 1983). Often joint production is possible in forestry such as the concomitant production of wildlife grazing and timber (Bostedt et al. 2003) and in agro-forestry schemes such as the production of Limba with Bananas for instance (Harou 1982). Multiple use management is an old concept for foresters (Gong 2002). The optimal forest rotation is a subject well debated in forestry economics (Hartmann 1976) and the reasoning is evolving with the new concept of sustainability (Erickson et al. 1999). How economic and environmental/forestry sustainability can be measured in order to be properly monitored and managed is also an important question for environmental economists.

Development and quality of economic growth

Economic development calls for higher per capita income via economic growth but a quality growth. The forestry sector economic growth for instance needs to provide a more sustainable natural environment, jobs in the poor rural areas, more amenities for forest recreation that will be paid for, stable local industries and the perpetual production of timber and biodiversity with their attached services. To attain sustainable development, it may be possible to reach both objective of economic and ecological improvement for instance in planting in mountainous region for both quality timber and protection against landslide and erosion. It may also be conceivable to use the forest ecosystems to obtain initial financial capital and let forests to re-growth later as was done in North America. However fragile ecosystems could be lost following this strategy of growth now and clean up later. High opportunity costs may be incurred in the future in following such a policy. Environmental economists attempt to quantify these trade-offs.

Public Goods, externalities, and the creation of new markets.

From the above, it is recommendable to design forestry sector development strategies based on better management of a broader portfolio of assets. The problem for many forest ecological functions and services is that they tend to have characteristics of public goods and externalities. The stocks of these environmental assets are then too small from society's perspective because of market or policy failures. Instruments need to be designed to correct these failures (Framstad 1996; Sterner 2003). Instruments need to be used for the forest products industry also (Brannlund and Kristrom 1997). The non excludability but rivalry, two characteristics of public goods, of common forests has been resolved in Europe after the middle age by privatizing some of them i.e. changing their ownership. Environmental economists are studying today the global common problems and research economic instruments, such the as Clean Development Mechanism (CDM), to damper the increasing CO_2 air concentration (Enzinger and Jeffs 2000). Forest ecosystems are important in that regard. A lot of global biodiversity are stoked in forest ecosystems also and economic instruments could be designed to protect it (Stenger and Normandin 2003). Examples of positive externalities, spillovers effects, are numerous in forestry. The environmental economists will have to design special conditions to create a market for these services, such as recreation for instance (Scarpa et al. 2000). Sometimes however the transactions costs to create that new market by defining new property rights may be too high to justify it. It may depend on such intangibles as the social fabric or the quality of the participatory processes. Institutional economics study in more details these aspects (North 1991). Contract theory and institutional choice (Grossman and Hart 1986), the economics of information (Stigler 1961) and principal-agent theory (Pattison Perry et al. 1998) are all theories of economics that are relevant in this context of internalizing environmental and forest related externalities.

Political economy

Policy failures may also be at the origin of environmental damages. The subsidies involved in the European Common Agriculture Policies have certainly increased the concentration of NO_x and SO_x in the ecosystems, forests and water in particular. Forestry incentives have in some cases reduced biodiversity (Stenger et al. 2003) The quality of water is particularly important (Matero 1996). Environmental economics help identify where the joint production of timber and water are economically more profitable than agriculture for instance in a particular area and at a given time. Public economics and the analysis of projects from society's point of view are practical tools used to analyze some of these problems duly incorporating environmental values when possible and through participative multicriteria decision making otherwise (Harou et al. 1996).

Environmental economists study all the above problems in the broad framework of the quality of economic growth duly considering the mix of three essential ecological, economic and social assets. Is it much different from what a thorough forest economist would do? It should not be.

Sustainable Forest Management in Europe

Helsinki Resolution H1 defines Sustainable Forest Management (SFM) as "the stewardship and use of forests and forest lands in a way, and at a rate, that maintains their biodiversity, productivity, regeneration capacity, vitality and their potential to fulfill, now and in the future, relevant ecological, economic and social functions, at local, national and global levels, and that does not cause damage to other ecosystems."

Such definition allows us to make an easy transition with what was said for the study of environmental economics in the context of sustainable development and now for forestry economics in the context of the definition of SFM provided by Helsinki Resolution H1. The joint production framework taking into account the three types of capitals is particularly appropriate for forestry understood broadly as in this definition. Fortunately, foresters benefit already from a good deal of environmental data to manage their forests sustainably. The application of these principles call for a multi-age, multi-species forest management minimising the entropy of the ecosystems, pro-sylva type if you want. Yet progress has still to be made on the social aspects. Rametsteiner (2000) describes modern forest management in these terms:

> *"Forest management encompasses the administrative, economic, legal, social and technical measures involved in the conservation and use of natural forests and forest plantations. Traditionally, forests have been primarily valued as a source of materials—timber and products such as resin and cork. Beyond this economic role, forests offer many other social and environmental benefits for the public. Over the past decades, forests have been recognized for their function in protecting biodiversity, local and regional climates, water and soil. In mountain areas, they serve the further purpose of avalanche control and protection against erosion. More recently, their importance in the fight against pollution and binding carbon oxide has grown. In general, forests serve several purposes at the same time. The underlying concept of management is called multipurpose, multifunctional or multiple-use forest management Due to the rising importance of ecological and social aspects in multiple-use forestry, the strict distinctions formerly drawn between production forests, protection forests and (nature) conservation area have become more blurred today."*

Such a definition again allows us to make an easy parallel between what was said for sustainable economic development. Most of the techniques, methods and approaches used and mentioned for environmental economics, revised national accounts, environmental valuation, economic instruments calibration to internalize forest externalities are relevant here. The difficulty in applying them to forestry is rather that all situations are locally and timely specific but the general economic principles hold for all of them. It is difficult to generalize an environmental valuation from a particular study to another area and at another time (Stenger 2000). Forestry economics is eminently location and time specific. This does not mean that results from one region cannot be used in another. However, the transfer of valuation studies for instance will have to be organized to be transparent and systematic in sharing the information from previous environmental economics results (Stenger 2003). A new network of excellence will be helpful in organizing that knowledge at the European level to facilitate its use and the relevance of its application. How could that knowledge be organized?

An Environment/Forestry Economics Research Framework

What we would like to suggest for co-ordinating or rather systematically organizing the results of research in such a vast field as environmental/forestry economics, is to propose a general framework encompassing many of the aspects of forestry/environmental economics we described in the two earlier sections. This framework is only one way to organize this type of research. The idea to use such a frame of reference to unite researchers from different disciplines and countries had been proposed by Harou and Essmann (1990) for a new IUFRO (Internal Union of Forestry Organizations) working group "Integrated Land Use and Forest Policies".

A similar framework is provided here to initiate a dialogue and collaboration in forestry/environment economics at the European level. It can be modified or started on another basis if desired. It is important to propose a framework sufficiently ample to allow for all the facets of the discipline as described earlier and at the same time provide an appropriate juxtaposition of works that facilitate decision making at the local, regional, national and European level. The proper organization of all the good research initiatives of the center of excellence should allow the understanding of the research complementarities and provide a whole that is superior to the sum of the individual research. It would also facilitate the dialogue with policy makers and help set priorities for research funding in this area by the European Commission in addition to the national priorities.

The proposal to initiate the elaboration of such a framework has been adapted from Markandya et al. (2002). The World Bank Institute used such a framework to train decision and policy makers in the intricacies of applied environmental economics for sustainable economic development. In that sense it is very applied and yet it can also be used to organize in the same fashion the more theoretical work in environmental/forestry economic research on which the applied work has to rest.

In Figure 1, the five boxes in the ellipse represent the more macro and sectoral aspects of forest economics and policies. The rest of the boxes cover the analysis of investments both public and private. The national forestry sector cannot be thought in isolation from the global context both economically and ecologically (Box 1). The effervescence surrounding the globalization of the economy was visible during the G8 summit few weeks ago. Likewise the Johannesburg summit or World Summit on Sustainable Development (WSSD), shows us that the global common had become a reality that need to be organized as it has been done at the more local levels since immemorial times (WBOED 2001). The same is true at the European level. The political and economic reality requires designing many policies considering the broader European context. Often the European context will change the evaluation that can be made of forestry programs (Harou 1987). The Box 1 factors have to be considered first since they will impact all the price signals in a national economy.

The second group of possible environment/forestry research is the one relating macroeconomic policy and environmental/forestry impact (Box 2). They have to be undertaken before the sectoral considerations because they will influence prices in all the sectors of a national economy. Monetary policy, fiscal and budgetary policies resulting from structural adjustment in many of the European economies in transition will have a direct impact on the forestry sector by increasing or decreasing deforestation for instance but also on acid rain impacts on forest ecosystems. In Box 2, one can also include the research aiming

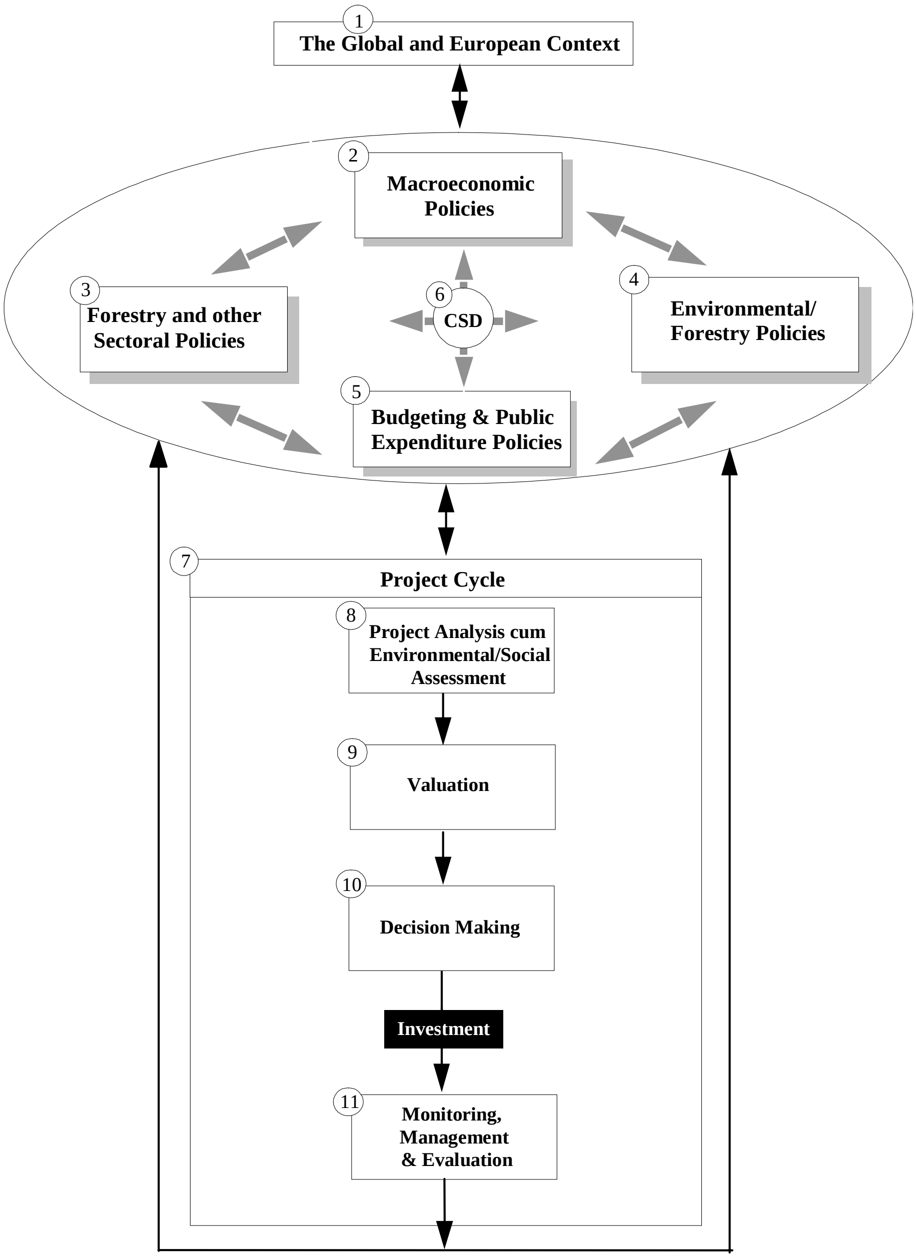

Figure 1. An Environment/Forestry Economics Research Framework.

at adjusting GDP for forestry net savings for instance or incorporating non-market outputs of forests into national accounting systems (Kristrom 1996; Peyron 1997; Peyron and Colnard 2002). General Equilibrium Modeling developed to assess environmental impacts of macro policies such as eco-taxes would also fit here as well (Alavalapati et al. 1997) as would the macro indicator of sustainable development (Rametsteiner 1999).

At the more sectoral level (Box 3), the partial equilibrium models of supply and demand of wood products and services would have their place here if the sectoral level were described broadly incorporating the different services and functions of the forests as discussed earlier. Environmental economists give often priorities to the demand side to propose economic instruments (Caviglia-Harris et al. 2003). Of particular importance would be all the research aiming at isolating the "right" prices of natural resources linked to forest ecosystems. That price should be corrected for market and policy imperfections typical of forestry environments. The economic and regulatory instruments proposed to reflect public goods and internalize important forestry externalities as well as other instruments of forest policies would have their place in Box 3. Voluntary instruments, such as eco-certification, are particularly appropriate for sustainable forestry (Costa and Ibanez 2002). Finally research on sectoral indicators of sustainability (Rametsteiner 2000; Adamowicz 2003) and would be gathered here also.

Box 4 would include the research on the main environmental issues identified in an Environmental strategy as well as the National Forestry Program advocated by the MCPFE. This research would help ranking the environmental/forestry issues by economic importance across all the sectors of the economy and submit priority actions to tackle them. The research on the methods of impact assessment of policies affecting directly or indirectly forestry (e.g. CAP policies) would also find their place here.

Finally Box 5 could collect the research aim at prioritizing national public investments from both Box 3 and 4. It is the budgeting related research presented in terms of public or welfare economics. This is best done using the program/ project or investment cycle to which we turn for the second part of the framework.

The different boxes considered at the level of the ellipse (1–5) could be termed the macro-environmental and economic perspective of sustainable development. They reflect in their ensemble the basis for a Sustainable Country Development Strategy (Box 6). The conditions set in the ellipse including the setting of a regulatory context and other policy instruments aiming at sustainability of the economy will directly affect investments, including forestry ones, made by private actors throughout the economy. We turn now to the analysis of these forestry investments with environmental and social impacts.

The second part (Box 7) of the framework is more easily understood as the public investment approach to forest/environment investments.

The classical Cost Benefit analysis complemented by an environmental and social assessment to duly consider the three pillars of economic sustainability at the program or project level is undertaken as a first step in the cycle (Box 8). Research helping to make these assessments more informative and relevant for the decision making process are needed. The same remark is even truer for the strategic environmental and social impact assessments of policies mentioned in Box 4 (Framstad 1996).

On the base of the environmental assessment, a valuation of non-market forests goods and services is needed to properly analyze forestry investments (Box 9). The evaluation research in this box could be the object of systematic storage of valuation results for organizing proper transfer of values when appropriate and times and budget does not allow otherwise (Desaigues and Stenger 2002).

In Box 10, we can include the research on stakeholders consultation as for instance reported in Ananda and Herath (2003). Not all the environmental and social impacts of forestry projects identified in the assessments will find their way in the economic cash flow of the investments. Then other indicators than monetary will be used together with the economic profitability indicators to make a multi-criteria decision (Tarp and Helles 1995). Participatory approaches need to be considered here at the project level but also as in Boxes 3 and 4 at a more macro level when priorities have to be ascertained. (MCPFE 2002).

Last but not least, monitoring of both projects and programs and policies (Box 11) should be the rules in forestry where the long duration of programs and investments allows us to predict with almost certitude that nothing will happen as we had envisioned at appraisal (Harou 1987). To manage this risk and uncertainty, our best bet is to monitor projects closely to propose early rectification. This research on monitoring will also allow homeostatic feedbacks to the investments and policies over time allowing reorienting them as we move along in their implementation.

The example of organization of environmental economic research for the forestry sector using an overall and coherent framework should be useful for estimating their joint pertinence for the improvement of management decisions and policies. It allows also to have the big picture in mind not only to set priorities among them, which we discussed briefly in the next version, but also to make clearer to the researchers the contribution they bring to the overall effort in organizing the sector and the economy toward sustainability. The information should be crucial to prepare the National Forestry Program in the perspective of economic sustainability.

European Priorities For Forestry Economics Research

Having proposed to organize the research in environmental/ forestry economics to foster synergies of efforts and better dissemination of research results, we now turn briefly, and to conclude, to the priorities for environmental economic research in Europe. The actions to undertake for following a sustainable path of economic development have been identified over the last decade and through the MCPFE[2] process and in light of the IPF/IFF[3] and now UNFF[4] global guidance. Clearly not all proposals are as relevant for environmental economic researchers. How to establish priorities is difficult to assess and will depend on the location specificity and the level of economic development of each country. However, at the European level, priorities could probably favor research on the European and global commons. Normally catalytic European funding with this objective will support locally motivated actions for which the immediate benefit is visible locally but with a synergy effect at the level of the continent and the planet. Ultimately, these priorities for research will have to be established by consensus and based on the vision we have for our continent. For some aspects the priorities can be established with the exact same economic tools used by the environmental economists.

It is also important to stress that research on forestry/environmental sustainability is essentially interdisciplinary and will require co-operation between disciplines at the European level. On this aspect it has become obvious that forest ecosystems management can only be seen in the frame of proper land planning (Helming and Wiggering 2003).

We would like to terminate by a point made on further research opportunities in a EU Commission report making the point on the first batch of research they finance on the socio-economic aspects of environmental change (EU 1998): “The ‘Europeanisation’ of Environmental, economic and social policy is a fundamental theme of great relevance. What precisely is meant by ‘Europeanisation’ is left deliberately vague. It covers a number of meanings, linking culture, to interpretations of the state, to nationhood, multi-nationhood, to concepts of sharing and redistributing.”

2 Ministerial Conference on the Protection of Forests in Europe
3 Intergovernmental Panel on forests/intergovernmental Forum on Forests
4 United Nations Forum on Forests

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Multifunctional Demands to Forestry – Societal Background, Evaluation Approaches and Adapted Inventory Methods for the Key Functions Protection, Production, Diversity and Recreation

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Abstract

Due to structural changes in the composition of the European and especially the German society two important tendencies have occurred over the past few years. First, in rural areas the increasing importance of a non-agricultural income has provoked a loss of identification with the use of lands – especially with the traditional forest use. Here the question of income alternatives in respect of harvesting is raised. On the other hand, particularly in urban areas, the interest in influencing political decision making concerning the sustainable use of lands – or even a protection against any human use – is stronger than ever before. Thus, proper approaches for the identification and valuation of the various societal needs considering forests are aimed to be developed as basis for a future-oriented forest management. This leads to the definition of different forest functions, which help to evaluate the costs and benefits of the above-mentioned societal demands. As a consequence, adapted inventory methods must be developed in order to provide the required information bases. The first part of the text focuses on the societal changes and their consequences for the future forest management with regard to an area-related integration or a segregation of forest functions. The second part of the article presents an approach for a producer-cost based evaluation of the expenses for a renouncement of harvesting. The developed potential supply curve may be a basis for private contracts between forest owners and nature conservationists. The third part introduces function-oriented inventory and planning methods taking into consideration the increasing complexity of the information required.

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1. Society, societal demands and structural changes

The international environment policy in the late 1980s and the 1990s led some European countries to a societal discourse on ecology and promoted participation in the decision-making process. These demands provoked conflicts concerning the existing forest management. Therefore, the representatives of the forestry tried to demonstrate to the society and the respective pressure groups that any societal concerns are unnecessary and accordingly, participation is not required because forests are already managed in a sustainable way. The term "sustainability" was placed in the centre of the forester's argumentation.

1.1 Sustainability in the consciousness of the society

Among others the Chair for Forest Policy and Forest History of the Technische Universität München conducted studies for six years referring to the term *sustainability* and its degree of familiarity in the society. The first results of different research institutions emphasised that the term *sustainable* has reached only between 13–25% of the population (Pauli 1999; Kuckarz and Grunenberg 2000) As a consequence, the questions to be posed regarding sustainability were differentiated according to the background of the interviewees: Suda and Helmle (2003) studied whether or not the people associate something with sustainability. If so the associations coming along with sustainability were analysed.

Of the 451 persons interviewed by telephone, 52% had no spontaneous connections with the term sustainability, 29% of the interviewees associated the term with their individual living conditions (*Lebenswelt*), 19% referred on general economic, ecological or social (political) aspects, like unemployment, stock exchange, bad economic situation etc. Forestry was mentioned only by two persons, the UN-Conference in Johannesburg (2002) by one. A linkage to the three pillars of sustainability – economy, ecology and society – did not exist in the consciousness of the interviewees. Moreover, the analysed opinions prove the missing understanding on the interdependences between forestry and sustainability. Thus, numerous contradictory statements within one answered questionnaire are possible (Suda and Helmle 2003).

As a result, the representatives of forestry do not reach the public with their core statement "forests are managed sustainably". According to Suda and Helmle (2003), a solution of this problem requires a communication strategy, establishing a relationship between sustainability and the individual living conditions (*Lebenswelt*) of the citizens. Besides, it should take into consideration that the term *forestry* is not yet known or is occupied with a negative image (Pauli 1999; Suda and Maier-Gampke 2000): The economic use of forests – i.e. the chain forest - forestry - timber – is either removed from the public mind or is not accepted as a necessary basis for the winning of renewable resources (Pauli 1999; Suda and Maier-Gampke 2000).

1.2 Demands of the society and social groups

Public demands against forest management are normally focussed on various types of recreation, thus not aiming to influence the methods of forest management itself. However, within the societal discourse concerning ecology, the protagonists of different environmental pressure groups have been actively participating in the forest-political arena from the mid-1990s on. This development has affected the certification debate and the initialisation of the Natura 2000 process, led to an increasing disclosure of protected areas and to a revision of the Nature Conservation Acts in different European regions. The term *biodiversity* has become one of the keywords.

Thereby conflicts were provoked in the forest-political arena, like the discussion on forest reserves (i.e. forest lands out of use) beyond the background of the forest certification. Within this debate, Greenpeace, WWF and other environmental groups, as well as forest associations, magazine manufacturers and retailer chains discussed criteria for a sustainable forest management. Two different certification systems – PEFC and FSC – resulted out of this process. The intended fusion to one predominant certification system failed because of the incompatible value systems. Moreover, the internal communication of the value-systems with the respective basis of each network made a reversion of the splitting into two competing systems impossible (Klins 2002; Walter and Klins 2003). Figure 1 provides information on the statements of forest owner associations and environmental pressure groups as representatives of the two systems. As the certification approaches were changed to recruit key players for a strategic alliance and to weaken the concurrence, these statements are no longer valid.

1.3 Structural changes concerning forestry

Structural changes affecting the proportions of agricultural and forest ownership in rural areas had severe effects on forest policy and forest land use – especially on aspects like harvesting and a negative impact on the maintenance of traditional (forest) values. One example is the closing of 12 877 farms in Germany in 1995–1997. At the same time, 4635 forest enterprises – most of them smaller than one hectare – were established, often without any agricultural background. Particularly for these small-scale forest owners an extensification of forest management activities in the future is rather likely. Especially in the case of urban forest owners, the distance between the residence and the forest lands is often very long, and thus

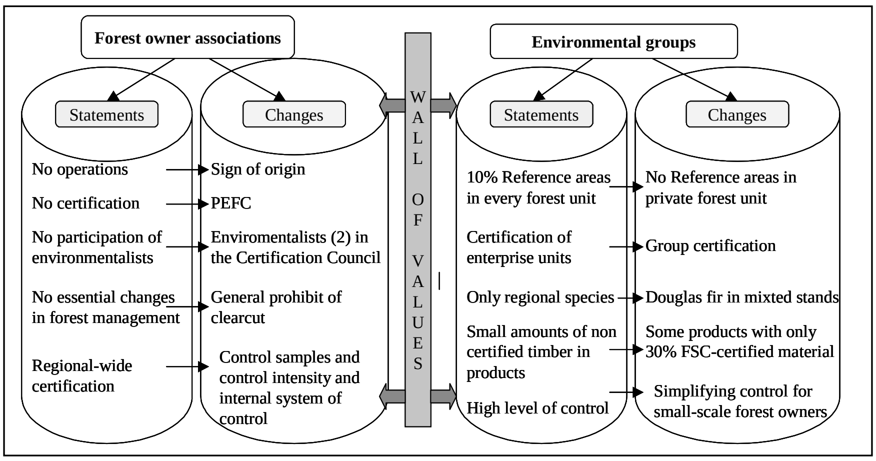

Figure 1. Changes and obstacles in the certification process (modified from Klins 2002).

regular management will be impossible. In addition, the increasing size of agricultural units in agroforestry enterprises will reduce the time to be spent for forest management activities. At the same time, traditional (forest) values like the so-called green savings bank, have lost their importance in favour of values like nature protection or the simple pride of owning land. As a result, the readiness to invest in equipment or knowledge is decreasing, along with the profitability (Suda and Ohrner 2001; Beck and Schaffner 2000). The most recent mission of the state forest administrations in co-operation with the local forest associations is to consult the small-scale forest owners about the mobilisation of their timber resources (Suda and Ohrner 2001).

1.4 The consequences of the demands: integration or segregation?

Assuming that the interests of the environmental pressure groups will continue to find entrance in the forest-political arena, e.g. by influencing norm amendments, the following question is to be answered: Which kind of management is the best reaction on this process? In this context, two types of management can be taken into consideration: the first one tends to integrate all demands and functions in the management of each stand, the second one, the segregation, tends to separate forest parcels according to different functions.

Segregation is a well known land-use principle: Until recently, agricultural uses were separated from forestry, transformation of whole landscapes into farmland by deforestation strengthened the segregation and even land use combinations like the forest meadow were reduced more and more. However, as to nature protection aspects in forestry, the idea of an integration of different functions into the management is strongly preferred. Here, the parallelism of use and protection "on each parcel in the whole area" is propagated. As a result the term *multifunctionality* was born and introduced within the forest-political arena, where an integration of all interests is considered possible or already carried out.

However, at least in Germany the problems going along with a multifunctional forest management have not been discussed critically until recently, as its consequences were not properly evaluated. Nevertheless, with the background of increasing budget restrictions in forestry and thus a decreasing willingness to pay for gratuitous services, the idea of segregation becomes more interesting. The most recent example of this change of mind is the debate on a contractually guaranteed payment for nature protection services.

While an integration of interests on the whole area can be regarded as a smouldering centre of conflicts, the segregation is the simpler solution because the participants can realize their ideas on "their" parcel of land. Here the provocative question should be raised: What will happen to forest lands if every claim is satisfied at any time by a function-related segregation of land-uses?

2. Approach for the monetary evaluation of societal demands

With regard to nature conservation, several environmental pressure groups in Germany intend to sacrifice the harvesting in considerable parts of the forest area. In the last decade, these groups considered 10% of the forest area as necessary to be protected, while the certification by the Forest Stewardship Council (FSC), Germany, requires in state forests 5% of the area as forest reserves. Corresponding to this development, the declaration of Natura-2000 areas by the European Union (Wagner 2000) also expresses a societal demand for natural forest reserves.

2.1 Forest reserves and producer prices

These tendencies and the very low profitability of forestry in Germany motivated Knoke and Moog (2003) to propose a methodological approach to evaluate the producer costs for the nature protection in forest reserves, where harvest benefits are per se excluded. Beyond the background of the so-called Contract-based Nature Protection (*Vertragsnaturschutz*) this approach could be seen as a basis for contract negotiations between private forest owners, demanding compensations for the abandonment of monetary benefit and the accredited nature protection administration or environmental pressure groups, who are requiring forest reserves.

Alternatively to adequate compensations from the state, prices similar to market prices can be calculated, which are inducing forest enterprises to establish forest reserves based on economic considerations. Bergen et al. (2002) report that Scandinavian households would pay between 5 and 35 euro per year to protect natural forests. Hence, the previously mentioned contracts can help to maximise the benefit of both nature conservationists and forest owners (e.g. Bergen et al. 2002; Moog and Brabänder 1994) and thus, show the way to a win-win-situation.

Here the essential question to be answered is: How to identify the compensation price, which can be seen as threshold for the profitability of forest reserves?

2.2 A potential supply function

As a first step, a potential supply function for forest reserves was developed. Basically, producer costs for forest reserves (similar to market prices) can be derived on the basis of the expected value of the non-harvested timber ensuing out of the formation of a forest reserve. This reflection indicated that producer costs for forest reserves are not constant in any stand type. If different stand types are arranged according to increasing opportunity-costs for forest reserves, step-function results, which can be seen as a potential supply curve. In Figure 2, the expected cash flows (net of logging costs) for the timber amount per unit of area to be harvested are depicted schematically for four stands. These expected cash flows are considered as the price of the "production" of forest reserves. This price multiplied with the corresponding area of the stand adds up to the total expected value of the potential timber harvest. Figure 2 shows that the opportunity-cost of renunciating timber harvests in the forest stand A1 would be the lowest ones per unit of area. In contrast to stand A1 the opportunity-cost per unit of area related with a forest reserve established in stand A4 are approximately eight times higher. The arrangement of the prices per unit of area for the four stands (Figure 2) forms the above mentioned step function, which here can be seen as a schematic supply curve for forest reserves. With increasing prices for forest reserves, the forest manager will allocate more forest land to this option.

When a specific price Y is offered for one unit of forest reserves, the forest manager will be able to assign an area of X units for the purpose of nature conservation. Area X is determined by the intersection point of the supply curve and the price line (Y), which in this case is a parallel to the X-axis. From an economic point of view, the establishment of forest reserves at an area of X units is advantageous for the forest enterprise. Its producer surplus can be derived from Figure 2 in the form of the grey area. The price achieved for forest reserves up to X units area is higher than the expected price for timber harvests in the stands A1 and A2, whose areas form X units. Thus, the forest manager simply adjusts the quantity of forest reserves to the price either given by a potential market or individually offered by forest reserve demanding groups. He will extend the supply of forest reserves until the marginal cost of the production of nature protection are equal with the price, which can be obtained for the last unit of forest reserve area.

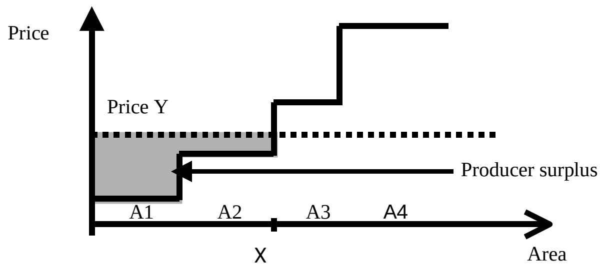

Figure 2. Potential supply function for forest reserves (Knoke and Moog 2003 in preparation).

Based on this theoretical consideration meanwhile a linear programming (LP) approach was employed to quantify the opportunity-cost of establishing a forest reserve. Integrated in this approach are several operational constraints, such as a maximum allowable cut, the minimum and maximum area for silvicultural activities and the demanded minimum cash flow in every period. In order to quantify the so-called shadow prices going along with forest reserves, also other technical plans like the operative plan were regarded: the shadow prices were calculated as the difference between the objective function of the operative plan with and without forest reserves. The opportunity-costs of forest reserves can be well reflected by the calculated shadow prices. Thus, the latter-ones form the supply curve for forest reserves.

2.3 Scientific prospect and research questions

Comparing the state-aided compensation for forest reserves with market-analogue mechanisms, the individual contract referring to nature conservancy seems to be most advantageous. Still there are open research needs to be discussed like the consequences of the privatisation of large forest areas especially in the eastern parts of Germany and also Eastern Europe for the further development of state-aided compensation and the willingness of pressure groups to pay for nature protection. Will there be a market-like development between the supply of potential forest reserves and a subsequent demand? Or will there be the necessity to apply governmental mechanism like laws in order to assure a minimum percentage of protected areas in each kind of forest property? Maybe first societal demands must be translated into operational, into functional information in order to reach at an area-related realization. Here adapted inventory-methods can help to manage this translation process by linking criteria and indicators for certain functions with the natural data basis.

3. Integration of multifunctionality into forest management planning

3.1 Basic conditions

When searching for an economic perspective for private and state forest enterprises, it should be emphasized that the perspectives of a future development are especially high, if forest enterprises are no longer limited to their role as primary producers but are widening their image with regard to a role as service providers (Brabänder 1995). Still, this necessary change of image is often regarded with scepticism. It is certainly justified to call attention to the hard experiences made in the past or the narrow legal margins. Yet an open discussion not only of timber production but also of the infrastructural achievement, which is often demanded as an extension of the services provided by forest enterprises, is essential in order to open up new yield potentials in the long run (Mantau et al. 1999). However, no general solution, which is equally suitable for every enterprise, can be achieved in this way. Instead, a wider differentiation including management types and the products aspired is to be expected.

3.2 Development of procedures

Beyond this background, forest sciences are obliged to develop ideas, which allow the implementation of a multi-functional forestry on the basis of reliable data regarding inventory, planning and control. Changes in the structure of growing stock and of the services rendered by forest enterprises are challenges, which cannot be met by simply modifying approved approaches. In the following, recently developed methods such as the type-oriented control sample, which is an integrative method of planning and inventory, and a concept for recording and evaluating forest recreation are introduced. Both procedures have the object to acquire sound and area-related basic data, which can be used a bases for decision-making and for securing multi-functional sustainability.

3.3 Type-oriented control sample

The vision of near-to-nature forests fulfilling multiple functions leads often to a horizontally and vertically intensively structured growing stock initiating natural regeneration. The classical inventory approaches in Germany are oriented on pure and homogenous even-aged stands and thus not yet suitable for this situation, which requires a more complex survey. Here both – the sustained yield control and the management planning – become equally hard. Therefore, operational strata-oriented sampling methods are used in the practice. However, they produce less detailed information due to the relatively large units of the findings (AG Forsteinrichtung 1997).

The type-oriented control sample is an approach, which provides statistically sound information on the growing stock in larger units such as districts or strata based on characteristic site classes. At the same time, it delivers stand-type-related information, which is essential for the planning and management execution at the stand level. The necessary terrestrial investigations are accomplished in a single work cycle by combining the classical control sample with the recording of stand type criteria in order to increase the flexibility of the data-employment: When a sample plot is recorded, universal stand type criteria for the surrounding reference area are documented, allowing a flexible assignation to stand types oriented on the needs of the forest enterprise. The variable typing supports an optimal use of the sample data with regard to changing targets, especially if different information is required during the planning period (in Germany usually 10 years). The stand types are the basis for a post-stratification of the recorded sample data. As a consequence, stand-information can be derived by a classification referring to the stand types. This is managed by recording type criteria for complete areas, mainly supported and realized by the use of aerial photographs in the course of the stand definition. Thus it is possible to breakdown the area-related and type-oriented inventory results to the single stand as unit for the silvicultural treatment. However, certain fuzziness with regard to the precision of the stand-related information must be tolerated in favour of an economic inventory and planning on the level of the forest enterprise

The terrestrial investigations are complemented by a photographic documentation of the type-specific situation focussing especially on assortment criteria of the growing stock with reference to an area-related optimisation of production and harvesting but providing also supplementary structural information with regard to the (bio-)diversity of the stock (Fürst 2004). Here also a link to the further processing and finishing in the forest wood chain is established by integrating intermediate and end-product criteria in the ensuing data analysis (Fürst and Seifert 2004).

The silvicultural management planning is based on type-oriented silvicultural treatment programmes, which also can be adapted to changing needs of the forest enterprise. Up to now these programmes are considering different age groups, site-conditions, the actual state and the target of the silvicultural development. The integration of multifunctional aspects e.g. with regard to nature protection, quality production or biodiversity can be managed by considering the actual structural state and the envisaged state for the most relevant stand types (e.g. oriented on the Natura 2000 criteria) in the type-oriented treatment programmes. During a final field trip, both the suitability of the programmed planning and the definition and of the type criteria are examined. If major discrepancies are found, an individual adaptation takes place. In straightforward silvicultural conditions, this last work step may be left out.

The blending of manifold data like forest biotope mapping, site mapping, remote sensing data etc. with the sample data is the link between the sample-plot based inventory and the management planning. It is based on the digital processing of plot and area-related data by means of a Geographical Information System. Figure 3 provides an overview on the sequence of the different steps of the type-oriented control sample.

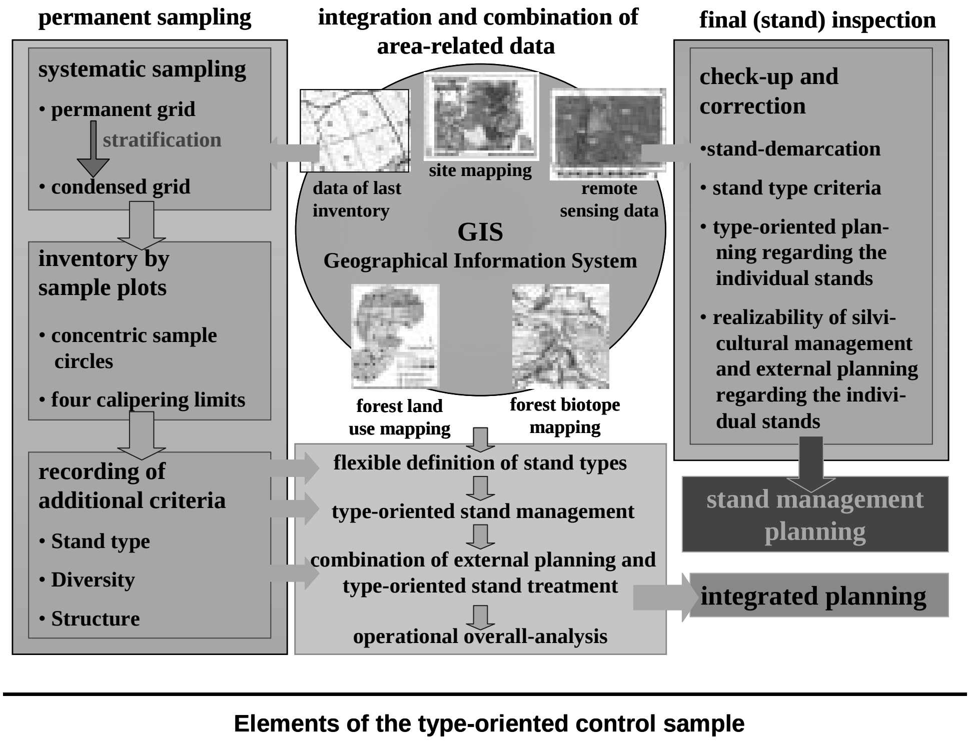

Figure 3. Type-oriented control sample (modified from Bitter and Merrem 1997).

One problem left is the integration of tending and harvesting on the one hand and the growth of the stocking volume on the other hand into the silvicultural treatment programmes. Here an adjustment of the inventory-data can be realized by a stable link to tree-growth models helping to actualise the stocking volume based on the growth simulation of type-characteristic stands (depending on the parameterisation for regional conditions: SILVA 2.2 or BWIN-Pro; Nagel et al. 2002; Pretzsch and Kahn 1996; Pretzsch et al. 2002). The implementation of a strictly type oriented simulation is a very economic way to use tree-growth models for the area-related up dating during a planning period. With regard to the tending/harvesting activities the data can be actualised by blending the data set of the timber-accountancy with the adjusted inventory data. Here the different levels of resolution – type-wise growth simulation and stand-wise, or sub-compartment-wise (in Germany) harvesting and tending accounting cause some problems resulting in a subsequent divergence of the precision of information on stand-type and single stand level until the end of the planning period. Nevertheless, the resulting error is relatively low compared with the use of non-adjusted inventory data. The stand-type oriented control sample and planning aims not to eliminate the silvicultural differences over large areas but should be considered as an efficient alternative to the classic inventory and planning systems, however with some constraints according to the precision of the single-stand related data.

3.4 Collection and evaluation of infrastructural services

Apart from wood production, it is one of the fundamental tasks of a multi-functional forest enterprise to supply infrastructural services. Here, forest enterprises need a detailed information basis on the real existing request of such services in order to realize an area-related optimisation of their offerings. Actually, there are often deficits concerning the quantification and evaluation of the inherently provided services and it is even more difficult to forecast their future importance beyond the background of increasing budget restrictions. Therefore it is necessary to develop methods for the spatial recording of the natural base, the time-scale related monitoring of the request and the integration into economic reference systems as decision basis for the forest management.

Regarding as an example the recreational function of forests, the quantitative data basis, which is available in Germany, is the result of the so-called forest function mapping. Here, conclusions on the individual functional level of forests are drawn from the recreational features and conditions of their surroundings and the local experiences of the forest administration. More detailed data on the request – e.g. the number of visitors – are scarcely collected. Of course it is possible to acquire statistically sound data on the number of far-off visitors (tourists) from the records of tourist associations. With regard to recreation-seekers from the same region, data can only be acquired by visitor counting. However, the various leisure activities and the diverse demands of different visitor groups to the forest as a recreational location are often neglected. Thus, possible conflict potentials reducing the recreational benefit of the individual visitor cannot be revealed.

A sampling concept taking care of both the spatial and temporal distribution of visitors and their special interests was developed for the Tharandt Forest (Tharandter Wald), which covers an area of 6600 ha in the commuter belt of Dresden (Saxony). First the entrances were typified by parameters like distance to the next build-up area, distance to the next car park and number of parking places, which according to a pre-test are the best predictors for the frequentation of an entrance. The recording of the number of visit events was carried out with the help of stationary and mobile reflective light barriers, whereas the number of cars was recorded by the use of hydraulic sensors. The stationary light barriers were utilized for a permanent yearlong recording at typical entrances, the mobile measuring facilities were used for a temporary recording at additional typical entrances over the whole model forest in order to validate the results. This approach helps to receive information on spatial fluctuations in the number of visits depending on the time of day and the season.

One problem left is the visit of groups, which cannot be registered exactly by the light barriers. Here a manual counting of visitors at each of the permanent barriers helps to detect the average number of groups and number of persons per group as basis for the correction of the light-barrier counting by a so-called personal factor.

The visitor's demands were documented within the scope of accompanying interviews in order to gain structural data on recreation-seekers as well as to obtain information on their individual preferences. Here, information on the type and frequency of the respective leisure activities can be obtained as well as e.g. information on the average walking or biking distance visitors can cover during their stay. The latter one forms the basis for the computing of the catchment areas of the entrances. Furthermore, special regard can be paid to the motivations and expectations of different groups with regard to conflicting interests of other groups.

Finally a Geographical Information System supports the area-related transfer of the information with regard to the differentiation of recreational zones according to the spatial and temporal intensity of their usage.

Figure 4 shows the results regarding the Tharandt Forest, where the total number of visits amounts to 730 000 per year. The results form the basis for a differentiated adaptation of the silvicultural management according to the local dominance of the function recreation. Thus, it is possible to compare the additional costs for an adapted silvicultural management with the benefit of the visitor evaluated by methods like the so-called *Reisekostenmethode* (travel-expenses method) or other economic approaches. In the long run, the results of the evaluation may constitute the basis for advanced product creation strategies and for better using the marketing potentials, which have been revealed.

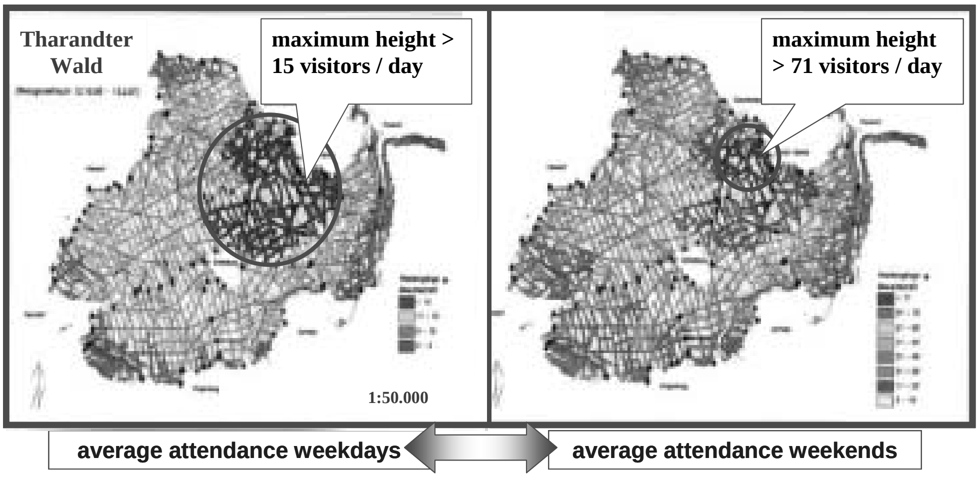

Figure 4. Map of "Tharandter Wald" (Schnell and Bitter 1997; Polster 2002).

4. Conclusions

As pointed out above, a massive communication deficit between (forest) decision makers and the society, here especially small scale land owners and environmental groups leads to rather woolly demands about the meaning and realisation of a sustainable and multifunctional use of forests. In this context, an amelioration of the communication strategies and a clear definition and valuation of the role and importance of forest systems against multifunctional demands would help to improve the mutual understanding. Besides the effects of integration or a segregation of different functions on the same area are calling for a proper evaluation. Wagner (2004) showed that some functions like the provision of drinking water with high quality and the production of a high timber volume can exclude each other. And even within one function – production of timber– the production of valuable single stems indicates stand structures deviating from those, which are necessary for bulk production. Thus, the definition of locally (stand-wise / compartment-wise) dominating functions (Ripken, 2004) can be seen as a first approach to consolidate the both concepts with regard to the total forest area of a specific region or a country.

The bases for the requested evaluation is a proper differentiation of single functions like production or protection and an economic valuation which should permit a comparison of the benefit or at least reveals the opportunity-costs of a distinct forest function against a "normal" forest management. The approach to estimate stand-(type)-wise the costs for the "production" of forest reserves helps to realize an area-related optimisation of the key functions timber-production and nature-protection with regard to an improved asking price for the negotiations with the nature-protectionists.

With regard to the wide range of forest functions a proper database is requested, which can only be delivered by adapted inventory methods. Here criteria indicating the suitability of the actual situation in respect of a special forest function as well as operational demands must be considered. Thus, on the one hand more complex information – e.g. focussing on the area-related diversity of forest systems or the frequentation of forests by recreation-seekers including their individual expectations – must be recorded. On the other hand, the costs for the inventory should be adapted to the increasing budget restrictions of forest enterprises. As a consequence, the inventory methods to be developed will have to tolerate less detailed information on the stand level in favour of broader area-related information on the level of large planning units.

Today's forestry is faced to multi-dimensional demands, which have led to the discussion on a multi-functional as well as sustainable use of natural resources. Here only a clear analysis of the actual natural and societal situation as well as an open communication of the economic consequences of such aims will help to preclude mutual misunderstandings and to find the path for a viable forestry of the 21st century.

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Economic Valuation of Non-Market Forest Benefits in Germany

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Abstract

Systematic efforts for valuing non market benefits of forests started not until the 1990s decade in Germany. Today, some 20 monetary valuation studies of environmental commodities have been undertaken in Germany about forestry issues. After a short survey of these, some empirical studies are presented which address the benefits forests yield for recreation, for the sequestration of atmospheric carbon, and finally, for the protection of biological diversity.

Open research questions concern, inter alia, the relation between forest composition and quality on the one hand, and recreation values on the other; the physical amount of carbon sequestered by forests and the dependency of the respective monetary values on the institutional framework of a carbon market; and the relation between single biodiversity protection measures and their respective valuation.

Keywords: monetary valuation; forest recreation; biological diversity; carbon sequestration

Introduction

The BFH Institute for Economics regards the integration of socio-economic aspects into sustainable forest management, and their translation into policy advice as one of its central research interests. Beyond problems associated with sustainable timber production (e.g. Dieter 1999, 2000), the institute focuses on methods for valuing environmental services of forests (see below), on participative approaches towards sustainable forestry (e.g. Elsasser 2002), but also on additional methods like e.g. Life Cycle Assessment (which may be considered an ‘economic’ method in that it helps improving resource use efficiency in the presence of external effects; e.g. Schweinle and Thoroe 2001; Schweinle 2002).

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The economic valuation of non-market benefits of forestry may be an important tool for supporting the sustainable use of forests. Many or even most of the outputs forestry produces have no prices since they are not traded in markets. In other cases, there are severe price distortions due to the presence of external effects, or due to government interventions fixing prices or quantities at certain levels, or creating artificial monopolies for specific products. In such a situation, valuing environmental (and other non-market) commodities can contribute to a more sustainable handling of forest resources, in that it makes values of such commodities comparable to the values of those products traded in the regular market.

Non-market forest benefit studies in Germany

German forestry has quite a long tradition in dealing with the sustainability principle in the presence of market failures for important environmental goods. However, it seems that the predominant response to the associated problems has been the attempt to circumvent them, rather than to search for a solution – theoretically by the development of doctrines which neglected the cost of capital in investment calculus ("Waldreinertragslehre"; cf. Möhring 2001), or by the ideology that the environmental services of forests generally followed in the wake of timber production ("Kielwassertheorie", Rupf 1960); and practically by truncating the regulatory power of markets by governmental intervention. Therefore, systematic efforts for valuing environmental benefits of forests started not until the 1990s decade in Germany. Until today, only some 20 forest related valuation studies have been conducted (see Table 1, except grey literature). But also beyond forestry, the experiences with the valuation of environmental commodities are still quite restricted. As an example, the number of contingent valuation studies currently at hand (which is the most widespread valuation method) was recently estimated at about 50(–100) for the whole German language area (Elsasser and Meyerhoff 2001). Internationally the number of such studies exceeded some 2000 already in 1995 (Carson et al. 1995).

As Table 1 shows, the existing studies already cover many of the different environmental services of forests. However, many of the available results originate from local case studies which are specific for the respective local circumstances. The approaches adopted there are often not suited for deriving generalised conclusions about forest values at aggregate level, simply because of the exorbitant data requirements they imply. Studies which do allow for such general conclusions only exist for the value of forest recreation, of the carbon sequestration service, and of the biodiversity protection service of forests in Germany. These will be presented in more detail below.

Recreation values of forests in Germany

Until now, by far most effort has been devoted to quantifying recreation values (see Table 1). Almost all of these studies have employed Contingent Valuation (CV), a valuation method which uses carefully developed interview procedures for estimating the willingness-to-pay of the relevant population for a given good or service.

Around 1990, several German research institutes had simultaneously started independent programmes for the valuation of recreation services of forests.[1] We initiated an early

1 Empirical results have later been presented by the Forestry Economics Institutes of Goettingen University (Bergen, Löwenstein, Luttmann, Schröder), of Freiburg University (Klein), and by the BFH Institute for Economics (Elsasser), some of these in cooperation with the Forest Research Institute of Rhineland-Palatinate.

Table 1. Non-market forest benefit studies in Germany.

Author/Date	Method	Scope	Location, federal state
landscape protection			*(no aggregation)*
Pfister 1991	hp	local	Niedernjesa, NS
recreation: holiday makers			*(ad hoc aggregation, Elsasser and Thoroe 1997)*
Bergen and Löwenstein 1992	tc	region	southern Harz, NS
Löwenstein 1994	cv, tc	region	southern Harz, NS
Luttmann and Schröder 1995	cv, tc	region	Lunebg.Heide, NS
Elsasser 1996	cv, tc	region	Pfalzerwald, RP
Gutow 2000	hp	region	Pfalzerwald, RP
recreation: day users			*(benefit transfer aggregation, Elsasser 2001)*
Klein and Elsasser 1994	cv	local	Flaesheim, NW
Elsasser 1996	cv, tc	region	Hamburg, HH
Elsasser 1996	cv, tc	region	Pfaelzerwald, RP
avalanche protection			*(no aggregation)*
Löwenstein 1995	cv	local	Hinterstein, BY
drinking water quality			*(no aggregation)*
Olschewski 1997	ac	local	Holdorf, HE
Gutow and Schröder 2000	mb	local	Kastellaun, RP
flood protection			*(no aggregation)*
Grottker 1999	ac	local	Vicht NW
microclimate protection			*(no aggregation)*
Löwenstein 2000	ac	local	Trier, RP
carbon sequestration			*(valued at aggregate level)*
Dieter and Elsasser 2002b	dc, ac, mb	region	Germany
biodiversity protection			*(valued at aggregate level; benefit transfer in prep.)*
Küpker and Elsasser in prep.	cv	region	Germany
Küpker in prep.	cv	region	SH; Germany
Meyerhoff in prep.	cv	region	Solling/Lunebg.Heide, NS

ac: alternative cost; dc: damage cost; cv: contingent valuation; hp: hedonic pricing; mb: market-based; tc: travel cost

coordination of the various valuation approaches already in the planning phase of these programmes, with the aim of developing a joint core methodology for all studies which would allow for a later comparison of findings. The result was a common base questionnaire applicable to day users as well as to holiday makers, which was jointly applied in all subsequent empirical CV studies on recreation. The valuation approach may be demonstrated using the day users' studies as an example, which were conducted in Hamburg (a densely populated city with about 5% forest coverage), in the Pfaelzerwald (a nature park of some 135,000 ha with about 75% forest coverage; for both see Elsasser 1996) and additionally in a small part of the green belt around the heavily industrialised Ruhr Basin (Klein and Elsasser 1994).

In all regions, data were collected by in-person on-site-interviews. Interviewers were positioned at the main entrances to the forests. In Hamburg and Pfaelzerwald, interview dates were distributed systematically over all months of one year, within months over all seven days (including holidays), and within days between 9 a.m. and 7 p.m. (the study in the Ruhr Basin employed a less elaborated sampling design, see Klein and Elsasser 1994). Since on-site sampling can lead to self-selectivity bias (due to different visit frequencies of the interview partners, as well as due to congestion intensities at the sampling sites varying over time), a data weighting scheme was developed, allowing to account for individual visit frequencies as well as for different use intensities at the sampling sites at different times.

Within the survey, forest visitors were categorized day users if they returned home the same day, and holiday makers otherwise. The contingent valuation question for day users was how much they would be willing to pay for an individual entrance ticket valid for one year and for all forests of their respective region. Respondents were reminded before that they already finance forest maintenance through their taxes. To help respondents answering, they were handed out a payment card which showed 27 different values and allowed them to specify a larger amount if they wished to.[2] After their initial answer to our valuation question, respondents were asked if they really would rather go without any forest visits than pay more if the hypothetical fee was higher than the amount they had stated. After this follow-up question, they got the opportunity to revise their initial bid. If their bid was zero, respondents were asked for their reasons. This allowed distinguishing genuine zero bids from protest votes. Travel and socio-economic data were obtained by additional questions. Interview refusees were asked to fill out a written questionnaire (containing travel and socio-economic questions only) and to return it by mail. Additionally, interviewers counted all visitors entering the forest at any of the sampling dates and noted some of their observable qualities (such as estimated age, sex, and group size). These data were later used for neutralising self-selectivity bias, and for comparisons between respondents and refusees.

The valuation results rather consistently revealed an average willingness-to-pay of about 50 € in all regions. Differences between sub-areas turned out to be more prominent, but did not change the estimates' order of magnitude; they were primarily attributable to different visitor compositions at the respective sites. Regression analyses confirmed the results' plausibility, as WTP was positively correlated to household income and visit frequency as expected, and negatively to household size, distance to forest, and furthermore, age of respondent. Moreover, significant differences between single sites remained after controlling for visitor composition, revealing higher mean recreation values in those forests located near cities. This points to the hypothesis that there may be value differences between forests attributable to forest composition. However, the knowledge about the specific reasons for such value differences between German forests is still very limited today.

In addition to the above, we conducted several CV validity investigations (on strategic behaviour, embedding (part-whole) effects, anchoring bias, answer refusal bias, on the influence of follow-up questions on valuation results, and on possible impacts of self-selection bias) which permitted an assessment of methodologically induced variations in the results. Beyond that, zonal TC analyses were calculated (in two versions to approach the inobservability problem with opportunity costs of time; Randall 1994). Both TC versions framed CV estimates and showed well comparable demand curves, hence not raising doubts about the CV estimates' validity (cf. Elsasser 1996, 1999).

Based on the results achieved so far, a benefit function transfer study was finally conducted in order to estimate aggregated forest recreation values for Germany as a whole (Elsasser 2001). This benefit function transfer basically combined the joined regression results of the available CV studies with an additional logit estimate of the forest visit probabilities in all German districts, which was derived by a separate population survey. Results showed that the aggregate willingness-to-pay of day users for forest recreation in Germany is within an order of magnitude

2 Due to several reasons, we preferred the payment card approach described here over the referendum approach recommended by the NOAA panel (Arrow et al. 1993). The main reason was that referendum type contingent valuations generally require larger samples, which would have prohibited many of our additional validity investigations described below. Moreover, with the referendum format it would have been impossible to account for the described self-selectivity of the respondents.

Figure 1. Regional distribution of forest recreation values per ha.

of 2.55 billion € per year. District averages of the willingness-to-pay ranged from about 25 €/year/ha to more than 25,000 €/year/ha, thus exhibiting very large regional differences. Figure 1 shows the corresponding regional distribution. In combination with regionalized values of other forest goods and services, such a map might e.g. support regional planning in identifying potential conflict regions or possible priority areas for specific services.

The value of carbon sequestration

The concern that global climate is threatened by increasing greenhouse gas concentrations in the atmosphere has provoked various counter activities by governments around the world. Economically, “air” and its composition has ceased to be a free good: the atmosphere’s limited capacity of serving as an emission sink for all sorts of production processes makes it a scarce production factor, which is of economic value. Managed forests sequester carbon dioxide in the course of their growth and thus contribute to a reduction of greenhouse gas concentration. Extent and economic value of this service are presently widely discussed.

In a recent study for Eurostat (the statistical service of the European Union), we estimated the overall annual economic value of carbon sequestration by Germany’s forests (Dieter and Elsasser 2002b). First, it turned out that even reliable quantity estimates of this service were not available. Although inventory results and yield table growth models exist for most tree species, these data are restricted to coarse wood. The available information on other biomass components (small wood, needles and leaves, roots, and herb layer) is already much more fragmentary, and information about soil carbon and its reaction to forest management practices is virtually absent at aggregate level. Therefore we had to proceed in two steps, estimating first the physical quantity of carbon storage in German forests (Dieter and Elsasser 2002a), and combining these with marginal unit values for carbon storage afterwards (Dieter and Elsasser 2002b).

In the first step, the available coarse wood inventory data were combined with appropriate conversion factors for carbon content in order to sum up the total coarse wood carbon stock in German forests. Next, gross annual stock changes (increment) was calculated, using a yield table approach for modelling coarse wood growth. Afterwards, several species specific expansion functions were successively developed and applied to account for the carbon content of small wood, needles, and roots, respectively; these are regressions based on literature meta-analyses. Net increment was finally obtained by subtracting harvests from gross increment. The estimates of physical carbon sequestration by German forests resulted in an annual net change of carbon stocks amounting to 54.38 $MtCO_2$/a; total carbon stocks were estimated at 3,961 $MtCO_2$ in the wood biomass, or 8,247 $MtCO_2$ including humus layer and soils.

The second step consisted in a combination of these quantity estimates with three different valuation approaches. The most obvious of these approaches was to collect prices for carbon sequestration which already are observable at the emerging carbon markets. These results were supplemented with two other methods, viz., the damage cost approach, and the damage avoidance cost (alternative cost) approach.

The development of markets for greenhouse gas (GHG) emissions is based on the emission limitations which are a consequence of the international negotiations following the Kyoto Protocol of 1997. The institutions necessary for guiding the GHG trade are still under development, and only few national governments have established regulatory programmes involving some possibility for trade as yet (the first being the United Kingdom and Denmark), which however differ in detail. In anticipation of possible governmental regulations, also some private firms have implemented internal emission trading prototypes to gain experience with trading and to detect possible problems. Rosenzweig et al. 2002) estimated that since 1996, approximately 65 trades with a minimum quantity of 1,000 t CO_2 equivalents have occurred worldwide, including trades of reductions as well as financial derivatives based on reductions (smaller trades and internal corporate trades are not included in this figure). Recently, regular emission trading has started at the London Stock Exchange. Some of the transactions effectuated so far have been published. The price span for transactions (until 2002) for which prices are available ranges from a little less than 1 to 10 €/tCO_2e; higher prices of up to 25 €/tCO_2e have been realised in the 1999 pilot phase of the BP internal trade. An average price might be assumed at approximately 5 €/tCO_2e, although recent developments at the London stock exchange seemed to tend towards somewhat higher average prices. This average also coincides well with price *expectations* of market participants (Natsource 2002). According to this investigation, an average price of just over 5 US$/$CO_2e$ was expected for the pre-Kyoto period (reference date June 30, 2005) in interviews with representatives of 35 companies operating in several industrialised countries.

Combining the quantity estimates presented above with the price estimates, the annual carbon sequestration service by German forests would have to be put in an order of magnitude between 54 mill. and 544 mill. €/year at prices between 1 and 10 €/tCO_2. The mean price estimate of 5 €/tCO_2 would accordingly give a value of about 271.9 mill. €/year.[3] Model estimates of emission avoidance costs available in the recent literature confirmed this order of magnitude if they assumed that global trade was permitted; for this case marginal

3 It has to be stressed that this is an estimate of the (global) welfare gain caused by the net carbon stock accumulation in Germany's forests. This estimate should not be confounded with the possible profits which might be realisable by German forestry enterprises under the specific accounting and trading rules which are currently being developed.

costs around 8 €/tCO_2 were reported (again under the emission limits imposed by the Kyoto protocol). However, marginal avoidance costs are quantity-dependent and increase with the amount of reduction. Between sectors as well as between countries, large differences in marginal avoidance costs exist. This is the reason why the extent to which emission trading (between sectors as well as between countries) will be allowed may significantly influence the 'price' a country will have to pay for reaching its reduction goal. The available model estimates illustrate how influential the flexibility of trading rules is on avoidance costs: If trade was restricted to the EU only, resulting costs (averaged across sectors and countries) were four times higher than under global trade; and under a no-trade assumption, costs were even 15 times higher (Mantzos 2000). Reported damage cost estimates – which are independent from politically fixed emission limits – were, on global average, in the same order of magnitude like market prices for CO_2, but turned out to be rather uncertain, partly due to limited knowledge about the possible consequences of the climate change, and partly due to various aggregation problems (aggregation across regions, across sectors and across time (discounting)). – But however prices will evolve in the future, it seems unlikely by now that the European GHG trading scheme to be developed will also accredit carbon sequestration by domestic forests. In this case, this service will therefore remain a public good outside the market sphere.

Biodiversity values

The third study to be presented here is part of an interdisciplinary research project on the biological diversity of forests in Germany, which is jointly elaborated by four institutes covering the fields of genetics, ecology, and economics.[4] The joint project aims at a better understanding of the mechanisms which influence genetic and species diversity within forest ecosystems, with specific emphasis on conditions in northern Germany. Within this frame, four studies address the influence of gene flows on the conservation of genetic diversity at population level; the relevance of autochtony for the stability of forest ecosystems; the dynamics of forest vascular plants at landscape scale; and finally, the socio-economic assessment of measures to protect forest biodiversity, including the monetary valuation of costs to the forest enterprises and of benefits to society. Costs are currently being researched by Gustav Küppers, and benefits by a PhD student, Markus Küpker (In prep.).

Methodologically, the benefit valuation study again applies Contingent Valuation in a questionnaire design comparable to the one described above. Two independent population surveys are conducted as oral household interviews, one addressing the population of Schleswig-Holstein (the northernmost federal state of Germany), the other the German population as a whole. In both surveys, the possible implementation of a biodiversity protection programme is described to the respondents, which has been specified in collaboration with the project partners from genetics and ecology and with forest practitioners (presently, some elements of this protection programme are also being intensively discussed in the policy domain). The programme consists of five measures: conversion of pure conifer stands into mixed or broadleaves stands; conservation of dead wood; reduction of game density; establishment of specific protection areas; and cross-linking of fragmented forest stands. Respondents are first asked how they rate the single elements of the programme, and if they would vote for or against the whole programme. Proponents are then asked how much

4 These are, respectively, the forest genetics institutes of Göttingen University and of the BFH, Hamburg University's Chair of World Forestry, and the BFH institute for economics.

they would be prepared to pay annually into a program specific fund aimed at the implementation of the programme; the fund includes a payback mechanism in order to be incentive compatible (that is, respondents are told that the programme will only be realised if enough money is collected, otherwise the money will be paid back). Objectors are asked the other way round, that is, for their willingness to contribute to a fund aimed at preventing the program. All groups are asked to value the programme at two regional levels, i.e. for Germany as a whole and for Schleswig-Holstein alone (with the question sequence inverted for half of the respondents). This construction will also be used to fathom methodological problems, specifically the relation between substitution and so-called "embedding effects" (cf. Kahneman and Knetsch 1992; Powe and Bateman 2003).

Pretest results reveal that out of thousand representatively sampled respondents, a majority (58% of the population) votes for the implementation of the programme in Germany, but there is also a significant number of objectors (13%); 29% are undecided. Proponents' mean willingness-to-pay is about 43 €/person/a, whereas objectors are prepared to pay some 6 €/person/a on average in order to prevent the programme (with a significant number of zero bids in the latter case). The population average is about 34 €/person/a, or less than 80% of the proponents' willingness-to-pay, thus showing how important an explicit inclusion of objectors' disutility can be for avoiding truncation biases. Aggregating the mean value to the population as a whole gives a total of around 1.4 bill. €/year for the implementation of this protection programme.

Conclusions

Summing up the valuation results available at aggregate level so far, it appears that a view restricted to market goods would severely underestimate the benefits offered by (German) forests. According to the national forest accounting results, the net value added of German forestry amounted to about 2 bill. € in 1999. The value of the biodiversity protection program described above is roughly in the same order of magnitude (1.4 bill. €/year), and the recreation value even above this estimate (2.55 bill. €/year). The value of carbon sequestration (0.3 bill. €/year) is still in the same order of magnitude as the additional products of forestry (which have a market value of 0.2 bill. €/year according to the forestry accounting results).[5]

With respect to the valuation of non-market services, several research questions are still open. Besides the fact that almost nothing is known about important other services in Germany at aggregate level (e.g. groundwater values), even for those services for which aggregate results are available the data base is rather weak. Moreover, many interdependencies between supply and demand are only scarcely understood. These concern, inter alia, the relation between forest composition and quality on the one hand, and recreation values on the other; the physical amount of carbon sequestered by forests (especially in their soils) and its dependency from forest management, and the relation between specific biodiversity protection measures and their respective valuation. Besides disciplinary research on methods, these issues will require a strong cooperation between natural and social scientists.

5 It should be kept in mind however that the results compared here refer to different valuation concepts (consumer's surplus in the case of biodiversity and recreation, and prices in the other cases).

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Perceived Property Rights – the Case of Regeneration Cuttings in Finland

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Abstract

Property regimes and rules describe the rights and duties related to the use of any forest resources. Property regimes are private, state, common, and non-property (open access) regimes. Property, liability, and inalienability rules describe rights and duties more in detail. It is important to recognize that property regimes and rules, whether based on law or social norms, are evolving in society.

The perceptions of property rights also vary among citizens. In other words, someone may consider a particular amount of a resource as his or her endowment. It is proposed here that individuals' perceived property rights can be explained by altruistic motives and ethical commitments. In the empirical part of this study a contingent valuation survey explored regeneration cuttings and especially dead and wildlife trees (DWT) left to enhance biodiversity. It is concluded that perceived property rights are important elements in understanding citizens' monetary valuations of nonmarket goods.

Keywords: property rights; contingent valuation; regeneration cuttings; biodiversity

1. Introduction

Contingent valuation (CV) of non-market environmental resources typically asks respondents willingness to pay (WTP) for an increase in a public good, such as forest conservation. At the same time, the valuation study makes – implicitly or explicitly – an assumption on property rights, namely that respondents have a duty to pay. However, property rights of public goods are frequently unambiguous (Plourde 1975; Bromley 1991). In fact, several non-market environmental resources are under open access so that no legal rights and duties are enforced. This gray area of the law is constantly under dispute. Moreover, from the perspective of welfare economics in general and CV in particular, taking into account the perceptions of

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property rights may be more important than that of legal rights (Mitchell and Carson 1989: 30). This is because individuals' evaluations are always based on perceptions, not on actual measures as such. In general, the recent struggles in forestry consider the question of who should pay the costs of biodiversity conservation.

Empirical studies on behavioral distributive justice have shown that people frequently have personal standards of fairness for setting commodity prices and wages (Kahneman et al. 1986; Colby 1995; Rabin 1998; Akerlof 1982; Akerlof and Yellen 1990). In the context of natural resources, Lockwood's (1999) study on conservation of Australian native forests measured property rights as personally perceived variables.

Generally speaking, it seems that very limited attention has been paid to property rights in the CV literature. Indeed, several valuation studies have varied the property rights so that either WTP or willingness to accept (WTA) compensation for the same resource has been explored (e.g. Brown and Gregory 1999). However, few studies have explored the perceptions of property right allocations. In this respect, Lockwood's (1999) study on the management of state-owned eucalypt forests in Australia is an exception. The main idea of that study, however, was to measure incommensurable preferences. Furthermore, debate on the assessment of existence values provides a fruitful background for analyzing property rights of nonmarket resources (Krutilla 1967; Weisbrod 1964; Plourde 1975; Randall and Stoll 1983; McConnel 1983; Boyle and Bishop 1987; Smith 1987). It is proposed in this paper that individuals' perceptions of property rights to environmental resources can be explained with the concepts of altruism and commitment.

The aim of the paper is, first, to describe the actual systems of property rights, that is, property regimes and property rules; second, to theoretically explain individuals' perceptions of property rights; and third, provide some preliminary results from an empirical study that measured the perceptions of property rights.

2. Theoretical framework

2.1 Property right regimes and rules

Property right regimes, on the one hand, define the parties which have rights to a particular resource and, on the other hand, describe the decision-making process related to those resources. Four property rights regimes include state property, private property, common property, and non-property (Randall 1975; Bromley 1991; Eggertson 1990[1]).

The private property is the most familiar property regime. Property rights belong to the individual, household, or corporation. These rights are called non-attenuated, e.g., they are transferable. However, individuals' rights under state and common property are typically non-transferable use rights, labeled usufruct rights. Under the state property regime, agencies may directly decide and manage the property. For example, the Forest and Park Service in Finland does planning and practical forestry operations in state forests. A common property represents private property for the group of co-owners. Common property regimes have frequently existed among indigenous societies, in the form of villages or tribes that have had collective decision-making structures.

No doubt, there are resources without an owner, for example the amenity values of forests. This case of non-property is conventionally labeled in environmental economics as an *open-access* (e.g., Tietenberg 1996). New benefits and resources, such as biodiversity, seem to be

1 Eggertson (1990) uses the term communal instead of common, and common instead of non-property

at first stage under open access. Landowners simply have a privilege to manage biodiversity according to their own preferences.

In addition to property right regimes describing the ownership of rights, the rights themselves can be defined in several ways. Three property right *rules* are called property rule, liability rule, and inalienability rule. Under property rule, one has to negotiate with the owner before using the resources in question. For example, a forest industrial company has to bargain with a forest owner in order to purchase timber. The forest owners' right to trees goes hand-in-hand with the notion that other persons do not have that right, but they have a duty that binds them not to cut trees. In Finland, trees, linchen, and soil in the woods belong to this category. They belong to forest owner under property rule. If anyone wants to use them he/she has to first make a contract with the forest owner.

Under liability ruler, another party is allowed to use the resources and a payment or compensation is determined ex post. This rule is frequently used, for instance, when new industrial plants have been constructed. Adverse effects, such as water pollution, have been compensated ex post to those suffering from such effects. The Finnish Forest Act of 1996 enforced the forest owners' duty to conserve certain habitats important to biodiversity. If economic losses due to such conservation are large enough, forest owners are compensated. However, compensations are not based on any voluntary negotiations between landowner and the authorities. In this sense, the Forest Act obeys the liability rule against forest owners' interest. In general liability rule is applied when it is not possible to negotiate before the action. That is why it is typically applied in the cases of unintentional actions. For instance, if there were damages to remaining trees on the neighbor's land due to felling the authorities would set the compensation that the forest owner, responsible to the felling, has to pay.

The third or so-called inalienability rule simply denies any action. The Forest Act of 1996 protects small habitats that do not cause large economic losses to the forest owner. No forestry actions concerning these habitats are allowed if they threaten the natural conditions of the place, nor are any compensations paid.

One important possibility is that there exist no rights or duties concerning the resource. Someone simply has the *privilege* to use the resource, i.e. he or she is free to use it, while at the same time others have no right to stop such usage. This is the typical case with externalities and public goods. For instance, in Finland anyone is free to collect non-wood forest products, such as berries and mushrooms. No one – including the forest owner – has the right to prevent the collection.

2.2 Explaining Perceived Property rights

Property rights described with property rules and regimes evolve in several ways in different societies. Depending on the unit of analysis, different explanations for this evolution can be provided, such as historical, cultural, economic or political explanations (Scott 1983; Waks 1996). Evident reasons for the changes in the allocation of property rights are changes in technology, wealth, scarcity, and preferences (Bromley 1991; Colby 1995; Bromley and Hodge 1990). The view of this paper is individual, reflecting the empirical inquiry on individuals' WTP for public goods.

Behavioral distributive justice regards how people actually feel about distributive justice — how they choose to divide resources (Rabin 1998). The issue has been studied in the context of monopoly pricing (Kahneman et al. 1986), labor market (Akerlof 1982; Akerlof and Yellen 1990), as well as various laboratory experiments (Andreoni 1995). Results show that individuals contribute to public goods more than can be explained by pure self-interest and they often retaliate against unfair treatment. Customers also seem to be willing to resist unfair firms even at positive costs (Kahneman et al. 1986). This fact indicates that people have committed themselves to a certain notion of justice (Stevens et al. 1993). On the other hand, in the context of the existence values it has been suggested that especially altruism is important (Krutilla 1967; Boyle and Bishop 1985; Johansson 1987). I suggest here that altruistic motives and commitments are also relevant in explaining the perceptions of property rights.

There is theoretical and empirical evidence that altruistic motives or so-called warm glow may affect the valuation of nonmarket goods (Andreoni 1990; Kahneman and Knetsch 1992; Johansson 1992; Brown et al. 1996). In particular, two forms of altruism can be defined. First, pure altruism is connected to the desire of an individual to increase the level of the public good (Andreoni 1989, 1990). Second, impure altruism assumes that individuals derive utility from *doing* good, not from the good itself or from an increase in the goods available to others (Olsen 1965; Becker 1974; Andreoni 1989, 1990). The point is that both types of altruism can be related to different groups of people and different uses of resources. For instance, someone may have altruistic feelings towards people recreating in the forests. Because timber fellings, occasionally, result in a lot of waste wood that is a nuisance to recreation, one may in an altruistic way argue that forest owners should have a duty to clean up the waste. In other words, those hiking in forests should have the right to walk on open trails. Conversely, forest recreation may cause trampling and erosion. If one feels altruism towards the forest owner, one may demand restrictions on recreation. For example, in Finland people are free to ride bicycles in private forests. Someone may think that this should be prohibited.

The idea of commitment entails that an individual behaves according to a norm (ethical, moral or social) that restricts his or her own behavior on the basis of some long-term orientation (Gundlach et al 1995; Berghäll 2003). Sen (1977) has provided, in a sense, an extreme definition of commitment:

> *"One way of defining commitment is in terms of a person choosing an act that he believes will yield a lower level of personal welfare to him than an alternative that is also available to him…One area in which the question of commitment is most important is that of the so-called public goods."*

An individual may not personally consider that a public good, e.g., nature conservation, benefits him or her but feels it is a moral duty to support the provision of the good. Generally speaking, an individual is motivated by, on the one hand, self-interest and, on the other hand, the welfare of society. This dichotomy can be described with a dual utilities model, where self and social interests produce irreducibly distinct preferences (Harsanyi 1955; Sen 1977; Margolis 1982; Edwards 1986; Blamey and Common 1999). Participation in voting or contributions to public goods are examples of this kind of behavior. In a CV context, people decide to allocate income between themselves and their social interests to do their fair share (Stevens et al. 1993).

People may have a commitment to "nature rights" (Stevens et al. 1991; Lockwood 1998, 1999; Spash, 2000). Individuals may argue that nonhuman entities or qualities, such as wildlife or biodiversity, have intrinsic value and a right to exist independent of any personal benefit. Spash and Hanley (1995) asked in a survey: "Do wild animals, plants, or ecosystems have the right to be protected regardless of what it costs society?" Results showed that one fifth of respondents answered affirmatively. In the same survey 67% of individuals agreed with the statement "biodiversity should be protected by law, and we shouldn't have to pay money to protect it." Especially, the latter statement clearly explores respondents' perceptions of rights and duties. It can be said that respondents' have acknowledged the-polluter-pays principle. An action threatening biodiversity is considered as pollution and resource owners, not the general public, have to protect biodiversity at their own cost. Unfortunately, the links between these statements and any property rules or regimes are not at all direct.

On the other hand, it is possible that commitments are attached to traditional land use. Traditional agriculture has a long history in many places. People may have ‘heritage’ value associated with particular agricultural activities, such as cattle grazing (Lockwood et al. 1994). Rekola et al. (2000) found a particular commitment to landowners’ property rights. In the context of the EU nature conservation program Natura 2000 network, Finnish people were heavily in favor of private property rights because the policy planning process was seen to abuse these rights. This commitment was even more frequent than the commitment to nature rights.

Non-industrial private forest owners (NIPFOs) have an especial role regarding the open access resources of forests. At the same time NIPFOs are managers of private goods, i.e., timber, and consumers of open access resources such as amenities and biodiversity benefits. The role of nontimber benefits, conventially labeled as in situ values, in NIPFOs’ timber harvesting behavior has been studied by Hartman (1976), Strang (1983), Tahvonen (1999), Koskela and Ollikainen (2001). The main result from these studies is that forest owners’ behavior is changed from the behavior which is based purely on timber values. In an extreme case, NIPFO may leave a stand unharvested. However, even if NIPFOs derive utility from in situ values their cutting behavior is not socially optimum. The double role of NIPFO certainly has an effect on their perceptions of property rights. It can be assumed that more important role the timber harvesting has in the NIPFO’s economy more committed to timber growing he/she is. Generally speaking, the share of NIPFOs is great in Finland, a fact that should be taken into account in the further analyses of perceived property rights.

The future generations are also a possible objective for commitments. Considering empirical studies, for example Spash (1998) has explored the greenhouse effect and the concerns for compensation to future generations. Based on answers to three statements, respondents were classified into categories of belief in compensation. One group is of greatest interest here. It believes that future generations have a right to compensation regardless of the relative welfare arguments, i.e. whether future generations are better-off in terms of welfare than the present. It seems plausible that this group is in favor of using some kind of a liability rule. Interestingly, this group does not deny the right of the current generation to use resources, despite the resulting greenhouse effects. Instead, the group insists that compensation has to be paid *ex post* to those suffering, that is future generations. Commitment to guarantee the level of timber production to the future generations is reflected in the principle of sustainability In practice the principle is applied in the Forest Act, which establishes a duty to regenerate forests after final felling.

3. Survey design

Typically, a CV survey describes a policy proposal that assumes compensations paid to resource owners. Often resource owners simply have privileges to utilize an open-access resource. Respectively, the respondent’s willingness to pay is asked. The point is that a privilege as a base of WTP is not without a dispute. A respondent may have a feeling that it is not correct to ask him or her to pay because there are no legally binding rights given upon an owner with regard to the resource. Instead, such a respondent may insist that the resource owner should take care of the resource without compensation. On these occasions people may, for instance, turn to interest groups, such environmental NGOs. NGOs may demand public authorities to clarify rights so that resource owners will have a duty to conserve biodiversity while managing his or her land so that nonowners do not have to pay compensations. In practice, after negotiations between different interest groups, some kind of a compromise has taken place most frequently.

The subject of the present study was biodiversity preservation in privately owned forests in Finland. Since the 1990s, one of the major challenges in forestry has been to provide sufficient environments for all species, especially for those living in old-growth forests. They require decaying wood, which has become scarce due to intensive commercial forestry. Environmental NGOs have demanded urgent changes in forest management practices and a considerable increase in conservation areas in Finland. As a response to this demand, a recommendation to leave some DWT in all regeneration areas, including clear-cut areas, was launched in the mid-1990's. DWT should be medium- or large- size trees and they should not be cut later on, but be left to decay (Ministry of Agriculture and Forestry 2000). Several recommendations concerning DWT have been published by various organizations. However, DWT are not included in the 1996 Forest Act. Therefore, they are not under legal regulation and there neither is there any economic compensation for landowners who leave DWT in regeneration areas.

The study evaluated respondents' perceptions of property rights using mail questionnaires. The main objective of the survey was to compare two measures for incommensurable preferences: paired comparisons and attitude statements (Rekola 2002). To develop the different questionnaires we had discussions with the representatives of stakeholders. A total of 1100 Finnish households were sampled from the Finnish Census register. A pilot survey was done in October, and the main survey was carried out in November 1998.

First, the questionnaire measured beliefs concerning forest regeneration cuttings as well as the frequency of observed cuttings. Next, the regulation and extension of forest regeneration were described and an information leaflet was provided where three alternative forest regeneration policies with graphic illustrations of cuttings were depicted. Alternative *A*, traditional clearcutting, consisted of having 0 DWT per hectare; alternative *B* was the status quo forest regeneration practice with 15 DWT; and alternative *C*, a more environmentally oriented method, called for leaving 35 DWT. The future cutting amount on a national level may be higher than the current one in all alternatives because fellings in the past have been lower than sustainable yield. To describe the effects on timber production, average potential national timber production under sustainable forest management was given at respectively 16%, 14%, and 12% more than current cuttings.

Next, statements concerning perceptions of property rights were presented (Appendix). A question of WTP was placed in the context of dichotomous choice and open-ended formats. Finally, in both samples the questionnaires requested information about respondents' socio-economic background.

4. Preliminary results

The number of responses was 503 (46%). Responses to the statements concerning the perceptions of property rights were interpreted as follows. If an answer to the first statement was any of options 1-3, a respondent agreed with the statement. This indicated that any changes from the alternative A should be compensated. This further indicates the property right level A. Statement 2 indicated whether the perceived property right is B or C. If the answer was that 60-100% of the costs should be covered by forest owners, the respondent was classified as belonging to the property right category C, otherwise as belonging to category B.

Results show that around 40% of respondents attached their perception of property rights to status quo, alternative *B* (Table 1). A similar number of respondents were classified to have the property right level *C*, whereas only 17% belonged to the property right category *A*.

Table 1. Perception of property rights.

	A	B	C
Levels of decayed and wildlife trees (*DWT*)	DWT=0	DWT=15	DWT=35
Number of respondents (%)	87 (17%)	212 (41%)	214 (42%)

5. Conclusions

The analysis of property rules and regimes shows that the structure of property rights may vary a lot. The essence from the point of view of valuation studies is that privileges as a base of WTP are not without dispute or legitimate protest. A respondent may insist that it is not correct to ask him to pay because the resource owner does not have legally binding rights to the resource. Even though these rights exist, people may insist on modifications, for example, because social values have changed. These kinds of responses are typically classified as protest answers in CV studies.

This paper reports preliminary results from a study that measured Finnish respondents' perceptions of property rights regarding the amount of dead and wildlife trees (DWT) left in regeneration cutting areas. Results showed that the current practice of DWT and respondents' perception of property rights are equal only to 40% of respondents. A substantial proportion of respondents considered that it is the forest owners' duty to carry the costs of increasing the amount of DWT. Moreover, results show that a minor portion of respondents consider the current conservation already unfair, for which and they demand compensation.

It is possible that people have answered strategically, i.e., urged forest owners to pay although they have actually felt that they themselves have a duty to pay. This is an issue to be explored in the future research. This study demonstrated that a prevalent assumption in contingent valuation studies, the use of status quo -property rights, is questionable. Finally, it is proposed that the inquiries concerning behavioral distributive justice, such as the perceptions of property rights of nonmarket resources may provide useful information for forest policy planning.

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Appendix 1. The measurement of perceptions of property rights

Statement 1

In my opinion, **the current practice** (alternative B) has already affected losses, such as costs to forest owners, that should have been compensated.	fully disagree	7 Ç	6 Ç	5 Ç	4 Ç	3 Ç	2 Ç	1 Ç	fully agree

Statement 2

How the taxpayers and forest owners should share the costs, in your opinion, if the current practice B is replaced by the alternative C.

	Forest owners share should be: %											
	0	10	20	30	40	50	60	70	80	90	100	
Taxpayers pay all	Ç	Ç	Ç	Ç	Ç	Ç	Ç	Ç	Ç	Ç	Ç	Forest owners pay all

Conceptions of Democracy of Key Informal Interest Groups in Finnish Forest Policy

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Abstract

The reform of Finnish forest policy in the 1990s has changed traditional Finnish (neo)corporatist policy-making into a multi-stakeholder process, which includes environmental NGOs and representatives of social issues. The purpose has been to enhance democracy and sustain legitimacy of the forest policy. The goal of this study is to identify the key commitments regarding democracy both in the formal documents related to forest policy and the key informal interest groups' positions concerning democracy.

The division into the categories of forest and environmental actors (with two sub-divisions for the latter, namely reformist and radical) was determined through subject positioning. The preliminary results suggest that forestry actors prefer parliamentary democracy, whereas environmental actors prefer wider public participation. A large range of quite similar expressions of democratic and counter-democratic codes were found in the data; each side identified with democratic codes and described the opposite side with counter-democratic codes.

Keywords: forest policy; interest group; value; democracy

1. Finnish forest policy in transition

Finnish forest policy originated in the late 1800s with the intention to end the wasting of the forest resources that had become increasingly important for the expanding forest industry. From the beginning, the principle of sustainability was adopted from Germany as a central guideline of forest policy, but in practice the legislation did not become effective before the

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1930s. The Finnish forest policy, similarly to a number of industrialised countries dependent on forest resources, was production-oriented until the 1990s when forest and forest conservation policies were reformed and further integrated.

Policies related to natural resources and the environment can be defined narrowly, referring to governmental processes only. A wider definition includes the political forces of business life and civil society, such as citizens, companies, unions, informal interest groups and organisations, all of which try to affect public decisions as well as the actions of other actors (Doyle and McEachern 1998). The latter definition provides a better picture of Finnish forest policy because the functional interest groups, especially the forest industry and the land-owners, have had a central role in policy-making. This practise, at times labelled as (neo-)corporatist (Palo 1993; Eriksson 1995; Ollonqvist 1998, 2002), gradually became a more pluralistic ("multi-stakeholder") process in the 1990s when major environmental NGOs and representatives of recreational use were allowed to enter into formal policy-making. The preparation of National Forest Programme (1999) involved relatively broad public participation, incorporating not only organisations but in principle any interested citizens. The most recent process, started with The Conservation Committee for the Protection of Southern Finland's Forests (Etelä-Suomen…2002), included no fewer than 16 different informal organisations and 10 formal organisations, as well as a number of non-voting specialists.

These broader forms of participation have, however, appeared to be unsatisfactory for many informal actors participating in policy-making. State officials may also be slightly confused about the recent rapid evolution in the guidelines of policy-making and the needs for future development; in general, the same qualms seems to exist in state level politics and other policy sectors. Internationally, there also seems to be a growing interest concerning the forms of public participation in forest policy (Public…2002).

For these reasons, it has become vital to analyse formal commitments to democratic participation in forest policy as well as key policy actors' conceptions of democracy. The objectives of this qualitative study are to:

1. Recognise the key commitments regarding democracy as stated in the formal documents related to forest policy and forest-related conservation policy, and
2. Identify the key informal interest groups' positions on democracy, and identify the major differences between these actors.

The study uses macro theories (normative theories of democracy) as well as interpretation theories (Figure 1). The study falls in the category of constructionist research, concerned with describing actors' own methods of ordering the world (Silverman 2001).

Section 3 analyses the expressions of democracy in the formal documents. Section 4 uses interview data to identify the major value positions and dichotomies through which the key actors perceive democratic decision-making. Prior to these, section 2 gives an overview of democracy-related theoretical thought and identifies some general qualities, which are then compared to the empirical results in the discussion of the last section.

2. Theories of democracy

The principal normative theories of democracy are used as background (Table 1); the ones chosen are somewhat established in many textbooks (e.g. Beitz 1989; Held 1996; Setälä 2003). Nevertheless, these partially contradictory theories do not provide a final, consistent picture of the contents of democracy. Some of the theories highlight the best consequences, while the others stress the predetermined values of democratic process or discussion and

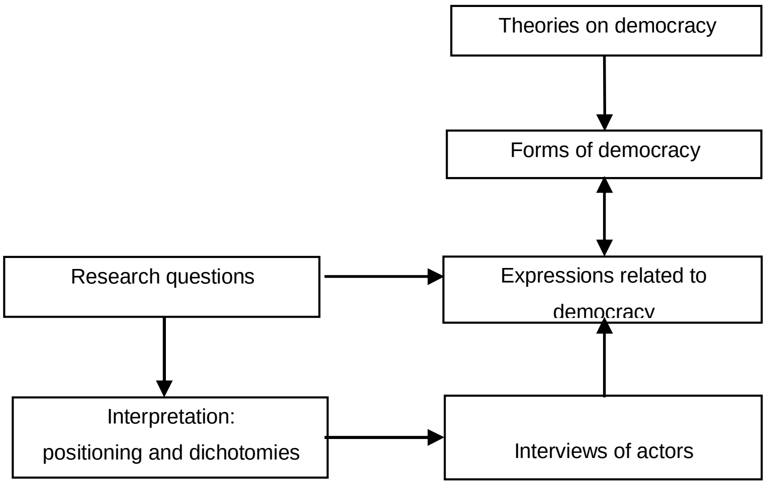

Figure 1. Structure of the study.

Table 1. Theories of democracy.

Consequentialistic	Will of the People (Buchanan & Tullock, Rae) Best results (Rousseau, Mill, Bentham, Riker, Popper) Development of citizens (Pateman, Barber)
Procedural / deontological	Political justice and equality (Beitz, May, Dahl)
Constructivistic / deontological	Communicative rationality (Habermas) Fair contract (Rawls)

deliberation during the process. Also, the emphasis on the macro and micro levels differs and affects their potential use in studies concerning actual political life.

The normative theories are general and abstract in nature, attempting to define norms and general conditions for democracy, independently of cultural or historical factors. To a degree this is a strong point, which provides leeway for an academic discourse and international comparisons, but it also limits the relevance of the theories. Some of the theories are philosophical constructions, speculating on situations and choices that do not exist in real life; thus their function is rather to give examples and stimulate thinking than to offer practical advice on reforming real life policies. Some of these theories are also noticeably idealistic and somewhat naïve.

Furthermore, from the point of view of their application, the major problem perhaps is that there is no consensus on an ethical view that should be used as a starting point. Theories of democracy are connected to political ideologies, such as liberalism, conservatism, socialism, and environmentalism (Heywood 1998), and when choosing a particular theory, the researcher is likely to commit him/herself to and promote certain ideological viewpoints instead of distancing him/herself from the actual political quarrels whose competing value positions cannot be solved by means of science. This problem of value-neutrality is of course present in almost all social and political scientific work and cannot be fully resolved at this point.

The problem stated above can, however, be at least partly avoided when the principal theories are applied simultaneously, recognising the common elements in the theories and using them to describe and compare different aspects of democracy in actual policies. Some shared qualities can be found in theories, although they are weighted differently in each theory, and not all included in the narrower theories. These are:

- Citizens' equal right to influence public decisions
  - in a certain geographical area (living area, nation's area) or
  - affecting one's own life (the quality and scale of effects must also be assessed)
- Majority rule; rights of minorities are safeguarded
- Possibility of influencing agenda-setting, principles of decision-making and outcome
- Preconditions: citizen and political rights, openness, transparency, access to knowledge, free media
- Political responsibility (accountability) through elections and availability of competitive alternatives
- Public discussion and deliberation
- Progress in citizens' activeness and abilities to deliberate

In the discussion, these different elements of democracy are compared to the declarations in the formal documents and the conceptions of democracy among the key informal interest groups.

3. Concept of democracy in formal documents and in National Forest Programme

Finnish forest policy-related documents from the 1990s contain only a few comments related to democracy. Still, we can find some signposts in international agreements, legislation, and programmes. One of the most influential statements was presented in the Forest Principles (1992) of the Rio process:

> *Governments should promote and provide opportunities for the participation of interested parties, including local communities and indigenous people, industries, labour, non-governmental organizations and individuals, forest dwellers and women, in the development, implementation and planning of national forest policies.*

Despite the non-binding nature of the convention, the Rio process has produced a kind of soft-law, which has affected the spirit of the times and committed policy-makers to the principles provided by the process (Rantala and Primmer 2003). The Rio principles of economic, ecological and social sustainability were adopted as the leading thoughts for forming the core of Finnish environmental programmes and forest legislation in the 1990s. Democracy-related ideas, however, were absent from formal forest policy documents prior to the National Forest Programme, which mobilised rather extensive participation in 1998-1999. Also, the Government's Programme for Sustainable Development (Hallituksen… 1998) appears to have given some additional inspiration for policy change.

At the same time, the Constitution of Finland was renewed and the participation of individuals was underlined: "government must support individuals' opportunities to participate in societal action" and "individuals must have opportunities to affect decision-making concerning their living environment". So far, the new constitution has not markedly affected the guidelines of forest-related policy-making, but in the future, it is supposed to increasingly guide forestry practises. Despite the new formulation of the status of direct participation, the societal atmosphere calls for even more radical reform in this area (Jyränki 2002). Some politicians in office have also expressed their concern over weakening legitimacy, especially regarding municipal level administration. Hence, a number of administrational reports and studies have been executed (Hallituksen…1998; Harisalo and Stenvall 2001; Kuule… 2001; Valtioneuvoston… 2002) and the new government has promised to advance possibilities for participation with the forthcoming National Democracy Project (Pääministeri… 2003).

Most formal documents consider participatory democracy as more supplementary than competitive in relation to parliamentary democracy. In forest sector practises, this relation appears to be somewhat fuzzy; furthermore, Finnish forest policy has a strong tradition of functional participation ("neo-corporatism"), supplemented recently with environmental NGOs ("a multi-stakeholder model"). All these forms of decision-making exist simultaneously in the National Forest Programme.

The growing number of participants and the search for a new policy procedure have been a source of frustration among policy actors. The following section analyses the major informal interest groups' value positioning concerning democracy, reflecting their perceptions regarding recent decision-making applications.

4. Key actors' conceptions of democracy: preliminary results

The purpose of the empirical part of the study is to explore key informal interest groups' democracy-related expressions concerning contemporary decision-making in forest and environmental policies. The study uses data consisting of eight semi-structured theme interviews, made in 2002. The interviewees played a central role in land-owner (n=2) or forest industry (n=2) organisations, or in environmental NGOs (n=4). The 18 questions, provided in advance, were general in nature in order to broadly identify the interviewees' forest and forest conservation policy-related values and beliefs. The expression 'democracy' was not explicitly mentioned in the questionnaire but appeared to be at the heart of the arguments; in particular the questions below generated large numbers of democracy-related arguments.

- How well functioning and fair do you consider current forest policy decision-making?
- How should the current decision-making system be changed?
- What means of influencing forest policy do you consider acceptable and which unacceptable?
- Who have the right to participate in forest policy decision-making?

The analysis is exploratory, however, using also Positioning Theory (Harré and Langenhove 1999; Törrönen 2001) and Jeffrey Alexander's (1998) concept in the interpretation. The former suggests searching for how the actors perceive "we" and the "others" or "they"; this indicates the different subject positions, including values and beliefs, which people either identify with, or which they perceive as opposite to their own position. Positioning is suggested to be a general quality of human psychology.

Alexander's concept identifies democratic and counter-democratic codes; his empirical study found these dichotomies to be widely pervasive in the American culture.

In the first stage of analysis, each interview was examined as a single unit of analysis. The analysis started with searching the subject positions and continued with recognising the expressions related to democracy and the opposite codes. All promising expressions were tabulated; furthermore, maintaining the same form, in which they existed in the transcriptions.

Table 2. Democratic codes accepted by all actors.

Democratic code	Counter-democratic code
Will of the People (will of the majority)	Personal opinions Vocal persons
Scientific knowledge Expert participation	One-sidedness Lying /manipulating
Transparency, openness	Non-transparent administration / organisations
Discussion	Influencing through media
Balanced decision-making	Distorted decision-making
Social sustainability	

Table 3. Democratic codes accepted by forestry actors.

Democratic code	Counter-democratic code
Parliamentary democracy	Direct action Voice voting
Mandate by members of organisation	Small number of members
Responsibility (economic responsibility justifies participation)	No responsibility for anyone Fixed ideological standpoint

In the second stage, similar expressions were collected in the groups. The comparison of the expressions in different interviews followed, and each group of expressions was condensed and the overlap was minimised. The similar arguments in different interviews were grouped in the summary tables and the expressions were named with appropriate short utterances, characteristic of the expressions (Tables 1–6).

The subject positions were easily found in the data; "we" and "they" were clearly expressed throughout the interviews. In the following analysis, the forest industry and land-owners are called forestry actors as a result of their fairly similar positioning. The environmental actors are generally defined as the opposite position and vice versa. Moreover, the environmental position can be divided into two sub-positions, namely reformist and radical, according to their arguments concerning the justified means available to "us" and "them". The latter division is not very clear in the interviews of forestry actors, who are more likely to group all environmental actors in the same position.

The analysis of dichotomies produced a large number of arguments, most of which are not possible to present in detail. In the following, the principal dichotomies are portrayed; some of them do not have a pair but it seems to be that an opposite alternative exists in their case, also.

The results suggest that the area shared by all actors (Table 2) is fairly broad, indeed. The will of the majority is at the core of all interviews, single and vocal opinions being the opposite code. Science and scientists are considered as a neutral source of information and their participation in decision-making is called for. Arguments related to transparency and openness are to a certain extent similar to science-related arguments, demanding truthfulness, but refer to administrative as well as to opposite organisations. Public discussion is

Table 4. Democratic codes accepted by all environmental actors.

Democratic code	Counter-democratic code
Influencing Government	Decisions by officials Parliament seldom changes suggestions presented by working groups
Wide participation Representation of NGOs in working groups	NGOs are not allowed to enter into decision-making Frames set before process Impossible to influence agenda
NGOs represent wide range of interests	Interest groups have no organised representatives

Table 5. Democratic codes accepted by reformist environmental actors.

Democratic code	Counter-democratic code
Influencing legislation in committees	Direct action (not criticised, not applied)
Expertise justifies representation of NGOs	

Table 6. Democratic codes accepted by radical environmental actors.

Democratic code	Counter-democratic code
Direct action, non-violent demonstrations Informing through media	Quasi-influencing in committee work
Common good	Broad rights of ownership

considered as a fair way to contribute democratic decisions. Balanced decision-making is called for by repeating almost word for word the formal forest policy: economic, ecological, and social sustainability should be taken into account in decisions; however, the social aspect in particular was seen missing in present policy. All actors agreed that the previous decision-making processes have been more or less distorted, leading to a situation that is far from balanced. At this point, it must be underlined that though the actors used the same arguments as a code of democracy, they made different reference; "we" as following the democratic code and "they" as acting in a counter-democratic way.

Parliamentary democracy was preferred among the forestry actors (Table 3), direct action and "voice voting" being the opposite codes. The actors emphasised the large number of members in their organisation, which has given them a mandate, and whose benefits, especially economic, they are responsible for defending. Those who do not have any economic benefit at stake were perceived to be irresponsible and constrained by non-negotiable ideological standpoints.

Several common characteristics were found in the arguments of all environmental actors (Table 4); they favoured markedly wider public participation than the forestry actors, as well as the broad participation of citizen organisations; however, the previous practises claimed to have a pre-set agenda and outcomes very strongly criticised. The actors were critical towards parliamentary democracy, which was seen as functioning poorly and giving the power to governmental administration. However, direct influencing to politicians was at the same time understood as democratic. The actors perceived themselves as a representatives of a wide array of different values generally accepted in the society, and of many non-organised interest groups, such as the owners of summer cottages, recreational users, and fishermen, hence they felt they had the legitimate right to participate in policy-making. Particularly the reformist environmental actors underlined that their expertise justifies participation, while the radical actors emphasised that they represent the common good and values of nature (Tables 5 and 6). The reformist actors did not strongly criticise the direct action applied by the radical actors, although they did not use it themselves. The radical actors saw the current property right as absurdly broad and falling in the category of the counter-democratic code.

5. Discussion

In general, arguments concerning the forms of democracy were found throughout the data. The arguments presented in the interviews covered almost the whole range of principal ideas in the normative theories of democracy (section 2). Possibly the only major idea almost missing in the interviews was that policy-making should promote citizens' activeness and develop their skills in deliberation and participation. The democracy-related ideas expressed in the formal documents were much narrower.

The arguments presented by those representing forestry position resembled models of pluralist democracy and competitive elitist democracy (Held 1996), while the environmental actors were generally closer to the participatory model. However, both positions had a number of similar arguments that were found in practically every interview but their conclusions as to, which actors are democratic and which are not, were exactly the opposite.

Some of the arguments in the interviews were used for describing and justifying the actions of organisations representing one's own position, while the "others'" depicted the codes of non-justified action. This argumentation structure found was rather similar to that found by Alexander (1998) in his wide empirical studies in the USA. Similarly, rationality and truthfulness represented a democratic code as well as deliberation through discussion and realistic argumentation. The single opinions of factions were also seen as an opposing code to the will of the majority.

The interesting question is, were the arguments presented in the interviews first order preferences, in other words, did they represent genuine value differences? Or were they second order preferences, i.e. were they instrumental in their nature, reflecting political resources which the actors possess and contributing to the pursuit of other goals? The latter assumption is usually made when studying political actors because rational actors are assumed to be instrumental in their nature. The general impression from the analysis of the interviews, however, is that both views seem to be valid: the good intentions of a democratic society and the instrumental views related to the game of politics seem to coexist in the forest policy argumentation.

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Ecosystem Functioning and Management

Modelling Biogeochemical Cycles in Forests: State of the Art and Perspectives

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Abstract

Biogeochemistry studies the movements of chemicals through ecosystems. The discipline has rapidly grown since its beginnings in 1840 with the work of Von Liebig on plant nutrition. At present, the cycles of water, carbon, nitrogen, phosphorus and many other elements are known in great detail. Based on this information, many biogeochemical models have been developed. Applications of the models in forestry have focused on problems relating to environmental change, especially concerning the effects of pollution, climate change and changes in management.

So the state of the art is that we have a fair understanding of biogeochemical cycles, have increasing quantities of data, and models that we can apply. However, progress can, and still needs, to be made. Various forest ecosystem types are poorly represented in the database, and long-term data are scarce. Many variables that are essential to biogeochemical modelling, like net primary productivity, have not been studied well, especially regarding their response to environmental change. Weaknesses in current biogeochemical modelling are lack of balance in the modelling of different components of the biogeochemical cycles, and insufficient quantification of the uncertainty of model outputs.

We suggest that the forest research community in general, and forest biogeochemical modelling in particular, might follow the example of the climate change research community in how they integrate research effort and identify areas where knowledge and predictive capability can be improved.

Keywords: carbon; water, nutrients; environmental change; prediction

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1. Introduction

Biogeochemistry is the scientific discipline that studies the movements of chemicals through ecosystems. As its name indicates, biogeochemistry is multidisciplinary, and elucidates the biological, geological, chemical and also physical processes by which chemicals are converted or transported. The discipline studies these ecosystem processes at a range of scales, from the individual stand of forest or other vegetation to the earth as a whole. Biogeochemistry is the extension of physiological ecology to larger systems and greater time spans.

We can trace back the beginnings of biogeochemistry to the first half of the 19th century, when Von Liebig first described the principles of plant nutrition (Von Liebig 1840). The importance of Von Liebig's findings concerning the demand of plants for nutrients was quickly recognised. We see an early example of this in the solution of the following problem, concerning the dying trees of Kensington Gardens. In the second half of the 19th century, many of the trees in Kensington Gardens, London, showed signs of premature death. The problem was discussed in Parliament but not readily resolved. In 1882, the geologist William Mathieu Williams pointed out that the trees were dying because Von Liebig's principles were not being followed. In his own words: "[The] reckless removal of leaves (…)" carried out only because "people might wet their feet" constitutes "agricultural vandalism" (Williams 1883). He pointed out that the removal of leaves had interrupted the natural cycle of events in which decomposing litter would replenish the nutrient pool in the soil. Williams understood biogeochemical cycling well: "Liebig taught (…) nitrogenous compounds are derived from the soil. (…) The possible atmospheric origin of some of the nitrogen is still under debate".

Since the 19th century, biogeochemical knowledge has increased greatly. At present, the cycles of water, carbon, nitrogen, phosphorus and many other elements are known in great detail. Much of this knowledge is qualitative: we know the nature of the processes involved but do not always know pool sizes and flux rates, or only know them for specific well-studied examples. In this paper, we focus on the biogeochemistry of forests. A simplified scheme, indicating the flows of carbon, water and nutrients between soil, trees and atmosphere is given in Figure 1. Strictly speaking, we cannot refer to these flows as "element cycling", as most of the atmospheric inputs to any forest ecosystem derive from sources outside the system, and most of the chemicals moving from soils to subsoil or atmosphere are transported out of the system. However, the internal fluxes are generally much larger than the fluxes across the system boundaries. We will thus use the term biogeochemical cycles to emphasize the continuous interaction between the different compartments of the forest ecosystem as depicted in Figure 1.

Biogeochemistry is an applied science. Many environmental problems, like air pollution and climate change, can be understood as human-induced perturbations of biogeochemical cycles. It is the task of biogeochemistry to quantify these disturbances and determine the effort required to maintain or return to equilibrium or some other desired state. However, chemical cycles are intricate and interlinked, and biogeochemical studies need to consider this complexity. This implies a need for models that quantify the processes and the factors that control them. In this paper, we will attempt to evaluate the present state of modelling biogeochemical cycles in forests. As no realistic model can be built without data of the real system to set model parameters and test model behaviour, we will begin by briefly reviewing data availability. Then we will discuss some examples of biogeochemical models, with emphasis on the representation of the mechanisms and controls that determine the rates at which biogeochemical processes proceed. We do not pretend to give a comprehensive overview of data and models: excellent books have been written on empirical and theoretical biogeochemistry (Schlesinger 1997; Smil 2002; Sverdrup and Stjernquist 2002; Waring and Running 1998). The main purpose of the examples we give is as a context in which to formulate suggestions for future progress in our field.

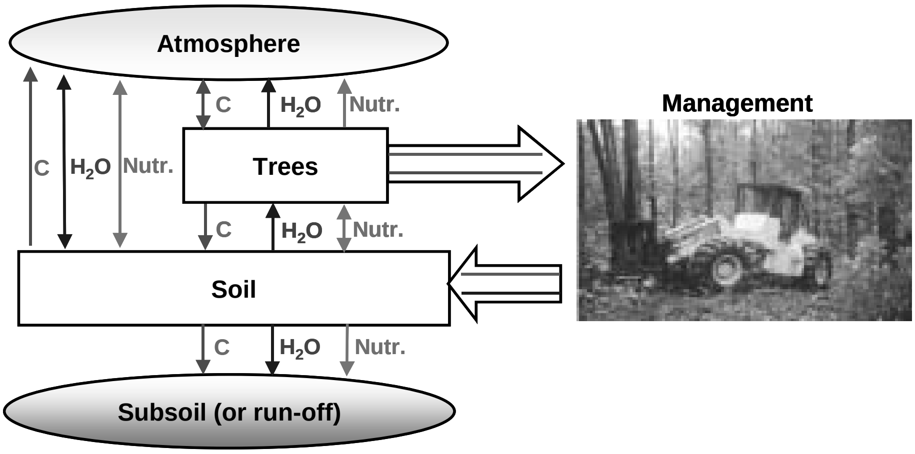

Figure 1. Forest biogeochemistry.

2. Data

At the global level, biogeochemical cycles are nearly closed: elements move between atmosphere, biosphere, lithosphere and oceans, with little gains or losses from the system. The fastest exchanges take place between the atmosphere and the other compartments. Because the atmosphere is a relatively well-mixed compartment, our data on its pools of chemicals are quite accurate. For example, the total global atmospheric pool of CO_2 is known at a precision (coefficient of variation because of sampling errors etc.) better than 1%, and the current global annual net flux of CO_2 into the atmosphere of 3.2 Pg C y^{-1} is known to about 3% (Houghton et al. 2001). Excellent summaries of estimated global biogeochemical rates and pool sizes are given by Reeburgh (1993–1997) and Schlesinger (1997).

Our knowledge of the biogeochemistry of individual forest ecosystems is less accurate. Forest ecosystems are open systems: their atmospheric compartments are coupled to those of the wider region through air movement, and their soil compartments are part of the regional hydrology. Human interference has made the systems even more open, as the example given above of litter raking in Kensington Gardens has made clear. Forestry itself is a diversion of biogeochemical fluxes for man's benefit. Although forest ecosystems are open systems, methods have been developed to quantify their annual carbon, water and nutrient budgets. Some studies have made use of catchments, where nutrient losses in surface water flowing out of a forest ecosystem can be monitored. Probably the most famous catchment studies are those that are being carried out since 1963 in the mixed forest at Hubbard Brook, New Hampshire (Likens et al. 1970; Bormann et al. 1977). Nutrient inputs from deposition and weathering have been estimated by balancing nutrient losses in outflowing water against changes in remaining pool sizes. Catchment methods are still in use (e.g. at Alptal, Switzerland (Schleppi et al. 1998)) but nutrient budgets are now also estimated by measurement of changes in nutrient concentrations in ground water, as in the semi-controlled conditions of the Solling experiment in Germany, where wet atmospheric deposition is regulated by the use of roofs installed in the forest (Cole and Rapp 1981).

The biogeochemical cycles operate at different speeds in different forest ecosystems. Cole and Rapp (1981) presented data on the average residence time in litter of organic matter, nitrogen and phosphorus, as measured in different forest ecosystems. The times were negatively correlated with temperature, varying from 230–350 years in boreal coniferous forest, to 15-18 years in temperate coniferous forest and 1–4 years in Mediterranean forest.

Many chemical flows in forests are tightly coupled to tree growth, so growth data collected for forest management may benefit biogeochemical studies. Standing stock volume can be estimated quite accurately. Phillips et al. (2000) found that stock volume in the forests of the South-eastern United States could be estimated to within 1.1%, with most of the small uncertainty due to sampling error. Changes in stock are more difficult to quantify, with errors reported for regional annual volume change up to 40% (Phillips et al. 2000). Measurement of forest net primary productivity is notoriously difficult, especially because of errors in below-ground production and turn-over (Clark et al. 2001). Recent developments in remote sensing and eddy covariance have improved the situation but these methods do still require ground testing (Clark et al. 2001; Moncrieff et al. 1996). Goulden et al. (1996) measured CO_2 and water vapour exchange for the Harvard Forest between 1990 and 1994, and their estimated errors were about 25%.

3. Modelling forest biogeochemistry

The varied data assembled on forest biogeochemistry have been used to develop process-based models. These models simulate the dynamics of chemical pools by representing the conversion and transport processes in mathematical equations or, more commonly, computer code. The models in turn have been used to help analyse the data, and to make predictions of forest behaviour under different conditions. A comprehensive forest biogeochemical model would include all the pools and fluxes shown in Figure 1, and also the chemical conversions within the different compartments that are not shown. A key feature of biogeochemical models is the stoichiometry of elements. Forest models have tended to focus mainly on C and N but, as for aquatic systems, P may require more attention. The limited flexibility in which the ratios of these elements can occur in various components of the system, and the difference in the ratios between components that are linked, impose considerable constraints on mass balances. Growth rates of organisms are therefore closely linked to element ratios. This particular research area, ecological stoichiometry, is developing rapidly and is informing the development of models (Sterner and Elser 2002).

The relevant processes in forest biogeochemical modelling can be categorised as follows: 1. Atmospheric dynamics, 2. Tree physiology, 3. Soil physics, 4. Soil chemistry, 5. Soil biology. In fact, most current biogeochemical models focus on no more than two or three of these categories. Often, the atmospheric processes are considered unaffected by the processes of gas exchange with forest vegetation and soil, so atmospheric dynamics are represented simply as input to the model. Exceptions exist, such as models that calculate atmospheric deposition as a function of vegetation characteristics (see e.g. (NEGTAP 2001)) Tree physiology is usually represented, at differing levels of complexity, but soil processes are represented less well (Jackson et al. 2000). Models tend to simulate only soil chemistry (common in models used in acid-rain research), soil physics (models used in hydrology or forest management) or soil biology (ecological studies on soil food webs, or soil biodiversity).

Process-based models are dynamic models, and they are used to study systems that undergo change. Forest biogeochemical models have been applied to forests subject to environmental change, such as air pollution, climate change or changes in management. The models are generally used to accomplish one of the following tasks: 1. Explain observed changes in forests, 2. Predict future change, 3. Evaluate management options. For each of these three tasks, we will give a few examples of application of biogeochemical models.

3.1 Application of models to explain observed changes in forests

In recent years, scientists have begun to recognise that the growth rate of many European forests has increased during the second half of the 20th century (Spiecker et al. 1996). In many regions, yield tables now underestimate growth. The EU-funded project RECOGNITION was set up to determine the cause of the growth acceleration. Preliminary analysis had identified climate change, increasing atmospheric CO_2 concentration and increased nitrogen deposition as possible causes. Four process-based biogeochemical models (EFIMOD, EFM, FinnFor and Q) were used to quantify the contribution of the different environmental factors to increased primary production in conifer forests at 22 sites across Europe (Van Oijen et al. 2004a). Figure 2 shows the results of the four models. There was large variation in the results of the four models, but all pointed to increased nitrogen deposition as the major cause of the changes in growth, except for the sites at the higher latitudes, where effects were smaller and dominated by changes in climate and CO_2 (Van Oijen et al. 2004b).

A second example concerns the possible role that forests can play in carbon sequestration to mitigate climate change. Recent years have seen an increased use of eddy covariance methods to quantify the sink strength of forests. In Europe, the main interest in this respect is in the capacity of young, recently planted forests to store carbon. Current research in Scotland

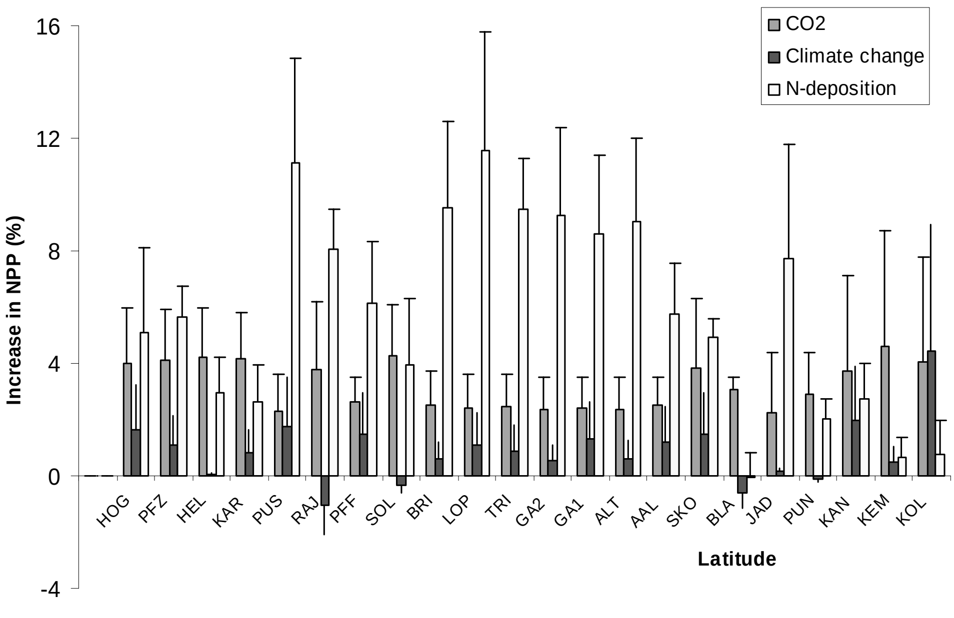

Figure 2. Effects of changes in [CO_2], climate and N-deposition on net primary productivity (NPP) of conifer forests at 22 sites across Europe between 1920 and 2000. The bars show average % change and standard errors for four process-based biogeochemical models (EFIMOD, EFM, FinnFor, Q). Sites are ordered from lowest latitude (HOG = Höglwald, southern Germany, 48.18 ° N) to highest (KOL = Kolari, northern Finland, 67.15 ° N). The effects of CO_2, climate and N were nearly additive, so overall change in NPP can be calculated as the sum of the three bars shown for each site. For more details, see Van Oijen et al (2004b).

attempts to quantify the changes in the carbon balance when ploughing peat land and afforesting it with conifers (Hargreaves et al. 2003). This is done in a chronosequence with sites at stages before and after the opening up of the peat. However, eddy covariance methods provide just one overall signal for an ecosystem, without differentiation into contributions from soil, trees or undergrowth. Application of the model C-FLOW has helped explain why the sink strength of young forest ecosystems was higher than expected, by attributing this to the temporarily strong growth of grasses and other herbs that bridge the time between peat ploughing and full forest canopy cover (Hargreaves et al. 2003; Milne et al. 2002).

3.2 Application of models to predict future changes in forests

In both projects described in the previous section, models were not only applied for explanation but also for prediction. In project RECOGNITION, the four biogeochemical models predicted for the 21st century that nitrogen deposition would become a far less important factor of environmental change than changes in climate and atmospheric CO_2, with especially strong effects of climate change at the most northerly sites. The peatland study using model C-FLOW is part of a larger British exercise aimed at predicting future changes in national stocks of carbon in soils and vegetation. Because of policy considerations, an essential element in that prediction is quantifying the uncertainty surrounding the expectations. This is accomplished by analysis of model sensitivity to changes in parameters and environmental scenarios.

The use of process-based models for prediction can be hampered by the fact that the models are rich in parameters that are poorly known, causing uncertainty in the outputs. However, using simpler statistical means of extrapolating current trends is dangerous if the system contains components that respond only slowly to environmental change, and thus may not be noticed in short term research (Van Oijen et al. 2004c). In many respects, the soil in forest ecosystems is such a slowly responding component, and biogeochemical models have shown how soil feedbacks may affect long term response of forests to environmental change (Jackson et al. 2000). One example of this is the study by Kirschbaum et al (1998), who showed that forest ecosystems that are "more open with respect to nutrient gains and losses are likely to be more responsive to increasing CO_2 concentration than systems where the amount of available nutrients is less variable". They concluded that ecosystems may become more nitrogen-efficient in the case of elevated CO_2, with a reduction in the loss of nitrogen to ground water. A similar, but even stronger conclusion was reached in a grassland modelling study by Cannell and Thornley (1998), who concluded that the relative response to elevated CO_2 may be smaller, in the short term, on nitrogen-poor sites, but is likely to be highest on those sites in the long term, precisely because of the increased nitrogen retention that Kirschbaum et al (1998) had found with their model. A third example of biogeochemical models pointing to the importance of soil processes for long term ecosystem response to environmental change is the study by Clein et al (2000), who showed by means of two models, GEM and TEM, that long term soil carbon storage crucially depends on whether recalcitrant soil organic matter has a fixed C/N ratio or not. Similar results were found in a modelling study on the effects of CO_2 and temperature on forests by McMurtrie et al (2001). As our final example of forest biogeochemical models suggesting long term effects that would have been difficult to predict from our generally short term observations, we note the work of Medlyn and Dewar (1996), who found that the long term response of tree growth rate to changes in CO_2, temperature, or both, depends on the strength of coupling between stem and foliage allocation.

3.3 Application of models to evaluate options for forest management

Forest management brings its own set of problems with it. There are complicated trade-offs to be assessed in forest management decision-making, which warrant the use of models. Rotation length and harvesting intensity need to be matched with the capacity of the soil to replenish removed nutrients. Thinning intensity needs to be commensurate with the demand for a robust wind-tolerant stand structure whilst permitting the penetration of radiation to seedlings below the canopy for regeneration. Solutions to the latter problem have recently been identified by means of modelling by Hale et al (2004). Management problems have become more complex in recent decades because of ongoing changes in the environment, and in the use of forests. Sustainable forest management has become the goal of many modelling studies, and the biogeochemistry of forests is an essential part of this. Examples, and thorough analysis, of forest management models are given by Bartelink and Mohren (in these proceedings).

4. Discussion

4.1 State of the art

The examples given in the two previous sections show that we have a fair understanding of biogeochemical cycles, have increasing quantities of data, and models that we can apply. However, progress can, and still needs, to be made. In this section, we list a number of possible deficiencies in current biogeochemical forest modelling, and suggest some ways forward. These deliberations are by nature somewhat subjective, but we hope they can stimulate further efforts at evaluating and improving our discipline.

4.1.1 Knowledge of flows and availability of data

Figure 1 represents forest biogeochemistry, in not too much detail. It only shows the major flows of carbon, water and nutrients between atmosphere, trees and soil. The flows that are represented are the ones commonly recognized and implemented in biogeochemical models. The reader will recognise photosynthesis, tree and soil respiration, transpiration, evaporation, nutrient uptake etc. Flows that are **not** represented (e.g. a direct flow of nutrients from atmosphere to trees) may be missing because they are non-existing or unimportant, or because they have been overlooked by modellers. We will briefly discuss these missing links first.

- *Direct uptake of nitrogen by trees from the atmosphere*. This process does in fact occur. For example, ammonia can be taken up by foliage if the concentration in leaves is lower than that in the atmosphere. Preliminary models of this phenomenon exist, albeit for grassland (Riedo et al. 2002).
- *Uptake of water by foliage*. This will generally be negligible compared to uptake by tree roots, but is important for mosses and lichens.
- *Uptake of carbon by tree roots*. Trees are capable of this. One example is the demonstrated uptake of HCO_3^- by willow roots (Vapaavuori and Pelkonen 1985). A second example is the uptake of organic nitrogen, including their carbon skeletons, demonstrated by Näsholm et al. (1998). These examples may be relatively unimportant for the tree carbon balance, but organic nitrogen uptake might contribute significantly to tree nutrient status.

- *Loss of water from tree roots to soil.* This process has been demonstrated (Pate and Dawson 1999), and may contribute to water transport from deep soil layers to drier upper layers (hydraulic lift). The process may be of special significance in multiple-species systems differing in rooting depth (Pate and Dawson 1999).
- *Atmospheric deposition of carbon to soils*. The flux of CO_2 is likely to be mostly upward, but deposition of larger carbon compounds (e.g. organic nitrogen, J.N. Cape, pers. comm. 2002) is possible.

These five examples emphasize the open nature of ecosystems: any kind of flow between two system components may be possible, albeit at widely differing magnitude. The key question thus is not "does a certain directed chemical flow exist?", but rather "what is its magnitude?". Unfortunately, the data we have on forest biogeochemistry are extensive but uneven. Long term data are scarce, except for forestry data which focus on tree height, diameter and volume, rather than on the different elemental flows. Moreover, the few more comprehensive biogeochemical datasets that we do have derive from a small number of research sites affiliated with major research institutes. Particularly lacking are data on mixed forests and on forests in Mediterranean climates. Further, our understandable focus on the problems of excess deposition of macronutrients onto forests may have obscured the limited availability of data on micronutrients.

The fact that our datasets have large gaps is perhaps best illustrated by the huge uncertainty in the estimates of the current European carbon sink. Whereas stand inventories suggest that the sink is 0.1 Gt C y^{-1}, studies with eddy covariance flux towers suggest 0.4 Gt C y^{-1} and inverse models (i.e. not biogeochemical models, but back-calculations from the spatiotemporal distribution of atmospheric CO_2 to the distribution of its terrestrial sources and sinks) suggest anything from 0.05 to 0.5 Gt C y^{-1} (Cannell 2002).

4.1.2 Information on mechanisms and controlling factors

Process-based biogeochemical models represent the ecosystem pools and flows of chemicals in some formal language, and add to this algorithms (rules) that determine the rates at which the various conversion and transport processes respond to changes in internal and external conditions. These rules are derived from knowledge of the underlying mechanisms and controlling factors. So, besides data, modellers need understanding of how the forest ecosystem works. However, this understanding is incomplete. A recent example of this was the study by Giardina and Ryan (2000), who showed that the model CENTURY overestimated the response of soil respiration to warming. The model apparently lacked negative feedback mechanisms that stabilised respiration rates. This issue is still debated, although more recent modelling studies (Ågren and Bosatta 2002) offer an explanation for the moderate soil warming response, by showing that soil quality (related to the fraction of easily combustable matter) may decrease quickly after warming, leaving a high proportion of temperature-resistant organic matter. This example not only shows that our understanding of biogeochemical mechanisms is still superficial, but also the useful interaction between modellers and empiricists.

As it is difficult to speculate about incompletely known mechanisms, we conclude this section with the following preliminary list of topics that may warrant closer investigation:

- Physiological adaptation of trees to environmental change
- Effect of root exudates on weathering rate
- Effect of mycorrhizae on nutrient uptake rates
- Genetic adaptation of the soil microbial population to environmental change

- Activity of soil microfauna
- Effect of soil texture on soil chemistry and dynamics of organic matter
- Role of macropores in drainage and leaching

Clearly, this list needs expansion before formulating a research agenda for forest biogeochemistry.

4.1.3 Quality of current process-based modelling of forest biogeochemistry

The deficiencies in availability of data and understanding of mechanisms, discussed in the previous two sections, affect the quality and applicability of our current suite of models. However, there are methodological issues in modelling that need to be resolved as well. The issue of lack of balance in the choice of biogeochemical processes that are represented has already been mentioned (Chap. 3). Further, many modelling studies still omit an analysis of the uncertainty of the outputs. Such an analysis needs to be performed to quantify the contributions to uncertainty from errors in both data and model (Van Oijen and Ewert 1999). Uncertainty analysis is not only important to the model user, who wants to know how reliable the model results are, but also to the model developer, who wishes to ensure that the model is neither oversimplified nor too complex to parameterise or test. The latter problem, overparameterisation of models, is common in biogeochemistry where the tendency exists to build models simply by coupling many existing models of system components (vegetation, soil chemistry, hydrology etc), while disregarding the statistical inadequacies of the resulting superstructure (see Sterman (1991) for a user-oriented discussion of simulation model complexity).

In various areas of environmental science, model comparison studies have been carried out to provide inventories of available models and to analyse the different approaches. However, such model comparisons have often been limited to listing structural differences between the models, with minimal comparison of their behaviour and without testing the models against data (Cramer et al. 1999; Kickert et al. 1999; Perruchoud and Fischlin 1995). Some comparisons go into greater depth (Homann et al. 2000). In the RECOGNITION project, mentioned before in Chapters 3.1 and 3.2, an effort was made to improve on this by testing all four used models against growth data (Van Oijen et al. 2004a) and by analysing how they differed in predicting environmental effects on the acquisition and efficiency of use of resources (light, water, nitrogen) by the forests (Van Oijen et al. 2004b).

4.2 Perspectives

We may conclude that forest biogeochemical modelling is an active and healthy research discipline. The field may benefit from greater investment in long term research sites (Brown et al. 2001), and a more geographically diverse distribution of them. The sites will be especially useful if they implement measurement programs that are less aimed at classical forest productivity indicators than on biogeochemical pools and processes. Continuous interaction between modellers and empiricists is desirable.

Modellers may need to pay more attention to the robustness and reliability of their models, by routinely testing models against data and by performing model uncertainty analyses. Bayesian approaches of quantifying probability distributions for parameters and outputs of forestry models are being explored (Green et al. 2000). Formal methods for assessing the uncertainty in the structure of environmental models are being developed as well (Beck 2002). Model robustness may benefit further from a change of attitude, where

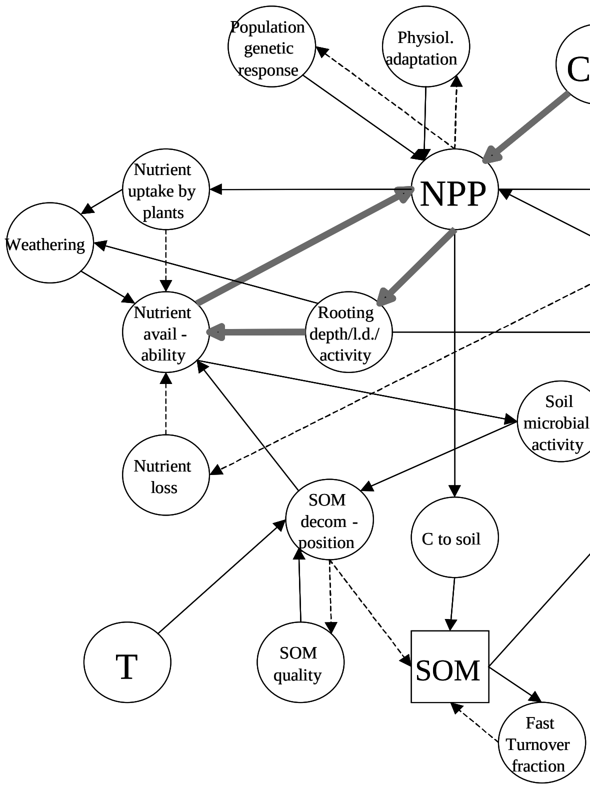

Figure 3. Feedbacks in ecosystems. Solid arrows indicate that an increase in a variable effects an increase in the target variable as well. Dashed arrows indicate a negative effect on the target variable. A feedback exists whenever two variables affect each other, directly or indirectly. Positive feedbacks are loops in which the number of negative effect-arrows is zero or an even number, and negative feedbacks have odd numbers of negative effects. The bold arrows show the example of a positive feedback: elevated CO_2 increases net primary production (NPP) which leads to increased rooting and therefore greater access to nutrients, thus further stimulating NPP.

modellers do not aim to include as many processes as they can formalise, but instead focus on including the essential feedbacks that operate in the studied system (Figure 3). Adding processes implies adding parameters, thereby increasing parameter-derived output uncertainty, often without markedly changing the average behaviour of the model. Adding feedbacks does tend to change model behaviour, but this reflects the behaviour of the real system (Sterman 2002). However, care needs to be taken that both positive and negative feedbacks are included, as the former increase model sensitivity to environmental change (Cox et al. 2000), whereas the latter tend to counteract them. Most importantly, feedbacks that operate at long time scales (such as through soil reservoirs, see Chap. 3.2) need to be included as they tend to dominate long term ecosystem behaviour.

We would like to end by commenting on the overall organisation of research in forest biogeochemistry. The research is still scattered among many different groups, who are not always well aware of what others are doing, with the result that gaps in knowledge persist like we have identified in this paper. The community of forest biogeochemical researchers, including both modellers and empiricists, might benefit from the example that has been set in climate change research, which is carried out in a well-coordinated way under the auspices of the Intergovernmental Panel on Climate Change (IPCC). IPCC Working groups show how research effort can be integrated even without institutional links between researchers. Moreover, the use of standardised scenarios for emissions and climate change, the fact that climate change and impact data are made freely available to all, and the regularly appearing reports on model comparison, all contribute to the current rapid progress in climate science that we are witnessing. A perhaps trimmed down version of such collaboration might benefit many research disciplines, including forest biogeochemistry. In such a context, the role of forest ecosystem modelling may be one that goes beyond providing tools for explanation and prediction, by also providing a nucleus around which research can be organised.

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New Aspects of Element Cycling and Forest Nutrition

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Abstract

Informative soil chemical parameters are needed as indicators to assess the ecological and nutritional status of forest soils. The amount and bond strength of exchangeable magnesium are examples of such indicators. Mg^{2+} is the most critical nutrient cation in forests of Central and Northern Europe due to its specific geochemical properties. It is very susceptible to leaching processes in acidifying forests. In the case of exchangeable potassium (K) extreme concentration gradients between aggregate surfaces and inner parts may control the K-uptake of trees and create K-deficiencies in South-West Germany. Apparently, the physical structure of forest soils affects the availability of chemicals for tree growth. A widely neglected fact is that the skeleton fraction of forest soils can contain a highly ecologically active nutrient ion pool. In skeleton-rich sites in the Black Forest Mountains, up to 80 % of the exchangeable Ca and Mg originate from the soil skeleton and not from fine earth. In light of these results, the traditional practice of only using homogenized fine earth for soil chemical analyses should be reconsidered.

Keywords: forest nutrition; naturally layered soil; soil skeleton

1. Introduction

The human race is performing four unintended large-scale experiments with forest soils in the northern hemisphere (Figure 1). These experiments can be characterized as follows:

- Greenhouse experiment
- Titration experiment
- Eutrophication experiment
- Soil-deforming experiment

We like to qualify forest ecosystems as "complex", which expresses the incompleteness of our knowledge regarding structures and processes. That means we are not able to give reliable

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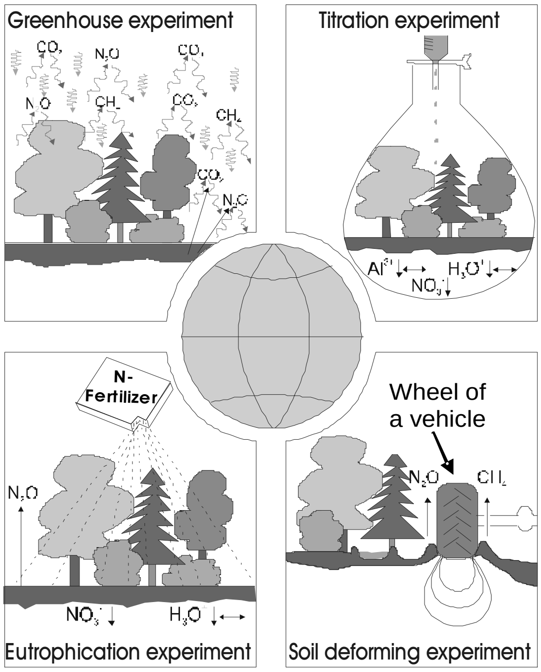

Figure 1. The human race is performing four "unintended large-scale experiments" with forest soils in the northern hemisphere.

forecasts about the consequences of these experiments. What we can do is simplify and focus on isolated aspects of forest ecosystem functioning. This paper deals with the impact of changing environmental conditions on soil chemical aspects of forest ecosystem functioning.

2. Greenhouse Experiment

Forest soils are sinks and/or sources of gases with greenhouse effects, especially CO_2. Both CO_2-fluxes are equilibrated at constant organic matter contents. As the humus content in forest soils is about twice as much as in agricultural sites, deforestation creates a CO_2 release into the atmosphere. The content of humus in forest soils is largely controlled by the temperature on a global scale. An increase of mean temperatures would therefore shift the equilibrium between humification and mineralisation towards higher mineralisation rates, thus provoking net releases of CO_2 from forest soils. Model calculations by Jenkinson et al. (1991) show that a temperature increase of 3 °C in the temperate zones in the next 60 years would yield an additional net release of CO_2 from soils amounting to ca. 30% of the current CO_2-release by combustion of fossil fuels.

3. Titration and Eutrophication Experiment

By emission of strong mineral acid precursors and the removal of these acids by rain, we unintentionally "titrate" (consume) the basicity of the ecosphere and we can observe results of

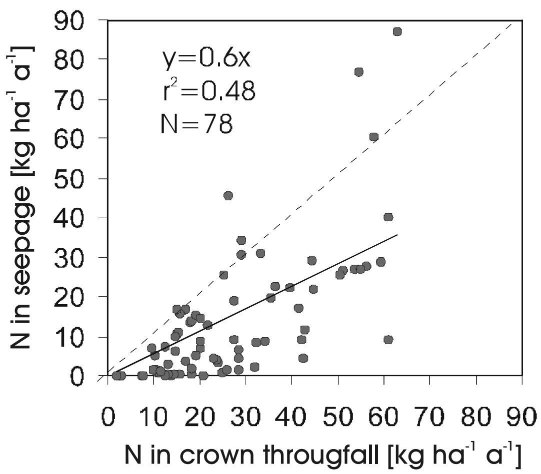

Figure 2. N-input in crown throughfall plotted against the N-outflow of European forest ecosystem studies (from Gundersen et al. 1998).

anthropogenically provoked neutralization and/or buffering reactions. The titration experiment increases natural acidification of soils in humid zones. However, the acidification by acid rain is triggered by anions of mobile strong mineral acids with a much higher coverage range as it is the case for acidity caused by organic acids (Reuss 1991). Therefore, acidification of subsoils and surface waters must be attributed to acid deposition in most cases.

The titration and eutrophication experiments overlap since the input of N by deposition is often coupled with an input of acidity. We have reason to consider natural forest ecosystems of the temperate zones as "N-limited-systems". Parent materials of soil formation normally do not contain N. The N pool cycling in these ecosystems originates from the very narrow "sluices" of symbiotic and non-symbiotic N-fixation. Therefore, the eutrophication experiment creates new conditions for forest ecosystems in the northern hemisphere.

The fact that at many European forest sites N-compounds appear in the transport media of soils (soil solution, soil atmosphere) can be seen as a symptom of an overloaded N-cycle in forest ecosystems. Figure 2 shows that many forest ecosystems are sources of N regarding the outflow.

3.1 Magnesium Depletion

Nitrate leaching pollutes the hydrosphere and reduces soil fertility. The law of electrical neutrality enforces the presence of cations in the soil solution if nitrate is leached. The cation with the lowest bond strength to negatively charged soil exchangers (e.g. clay minerals, humus) will preferably be mobilised and go with the nitrate. Due to its stable hydratation and the consequently low bond strength this will be Mg, as long as the supply of exchangeable Mg is not completely depleted. The GAPON-coefficients (k_G), which may be seen as a relative measure of bond strength, indicate that the relative bond strength of Mg decreases drastically with increasing soil acidification. In Figure 3, the acidification gradient is expressed as increasing (Al+Fe)-saturation of the exchanger in mineral horizons with a low C-content (Hildebrand 1994).

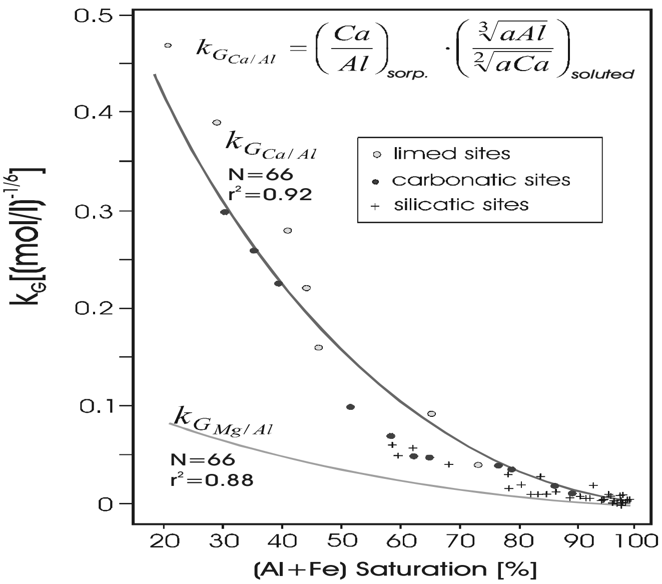

Figure 3. GAPON-coefficients (k_G of Mg/Al and Ca/Al) plotted against (Al+Fe) saturation of the exchanger at different sites in South-West Germany (from Hildebrand 1994).

In an ecosystem study of matter budgets in a spruce stand on moraine deposits in South-West-Germany we can observe the Mg-depletion triggered by nitrate export "in action". The depletion curves of exchangeable basic cations in Figure 4 are obtained (Lukes et al. 1996) by updating the actual K, Ca and Mg-exportation in the rooted zone (0–60 cm) with the seepage by a first-order reaction model and assuming optimistically a constant K, Ca and-Mg-input of 0.25 $kmol_c\ ha^{-1}\ a^{-1}$ by weathering (assigned to K, Ca and Mg according to their molar ratio in the parent material).

We see that the supply of exchangeable Mg will drop below 10% of the actual supply within 200 years if the rate of nitrate exportation with the seepage water continues and Mg is mobilized according to its decreasing supply at a constant rate (first order model). It is clear that such a simple model is not suitable for reliable forecasts of ion pools for a couple of hundred years. However, it shows that the actual rate of Mg exportation is very high and the site is outside a steady state of matter budgets. If we consider the very low supply of exchangeable magnesium in silicatic forest soils as monitored by the German Forest Soil Survey Program (Figure 5), such a depletion of exchangeable basic cations is very likely in the past decades.

3.2 The Selective Depletion of Nutrient Cations on Soil Aggregate Surfaces

Roots and mycorrhizal hyphae prefer growing on soil aggregate surfaces and do not penetrate deeply the soil matrix (Schack-Kirchner et al. 2000). Because soil aggregate surfaces can be selectively depleted of water soluble and/or exchangeable basic cations (Hildebrand 1994), nutritional potentials of the "real" rhizosphere are overestimated by bulk soil analysis. This particularly applies for potassium as percolation experiments with un-disturbed and homogenized soil samples showed (Hildebrand 1991, 1994).

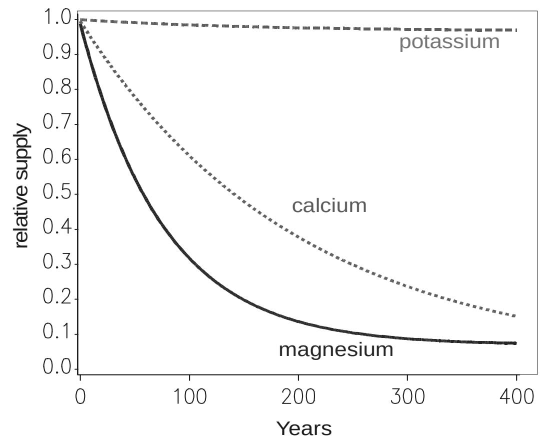

Figure 4. Depletion curves of exchangeable basic cations triggered by nitrate exportation assuming a first order reaction model (spruce stand on moraine deposits in South Germany (from Lukes et al. 1996).

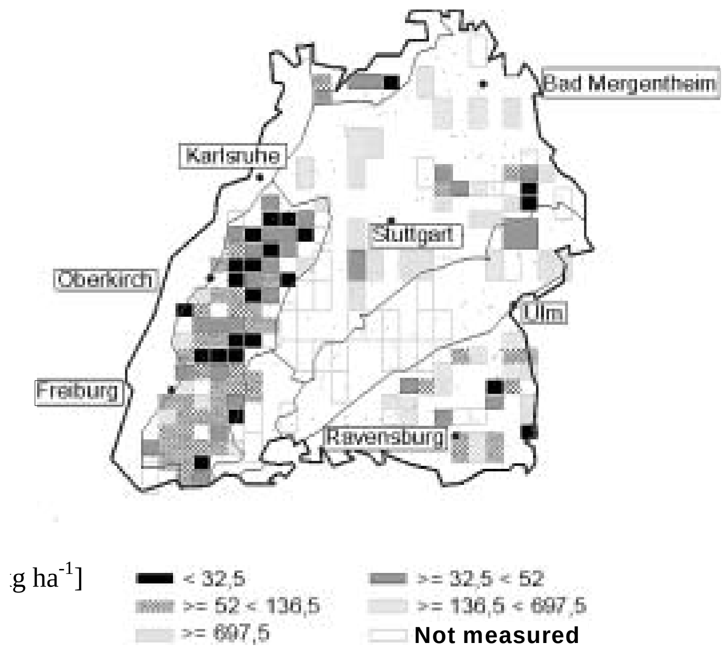

Figure 5. Supply of exchangeable magnesium in forest soils of Baden-Württemberg monitored by the German Forest Soil Survey Program; soil depth 0-60 cm (from Buberl et al. 1994).

Widespread K-deficiencies in spruce stands on moraine deposits in South-Germany as described by Zöttl and Hüttl (1985) could be explained by extreme-K-depletion of soil aggregate surfaces rather than by the bulk stock of exchangeable K. The effect of K-depletion on root accessible aggregate surfaces is non-linear since the bonding strength increases exponentially with decreasing K-saturation. Figure 6 shows that root accessible aggregate surfaces are characterized by very high K-bond-strengths in particular. The reason for such structure-dependent disequilibria in forest soils can be attributed to low biogenic aggregation formation rates, since such disequilibria were weaker or missing in previously limed soils (Hildebrand 1991).

3.3 The Importance of the Skeletal Fraction for Bio- and Geochemical Cycles

However, symptoms of deficiency are rather seldom despite the dramatic depletion of exchangeable basic cations in the mountainous regions of Central and Northern Europe. Furthermore, forest growth is generally higher than it was in the past. This creates a "deficit of plausibility" for soil chemists because even extremely low base saturations (base stocks <5%, Al-saturation >95%) seem to have no ecological consequences. This contradiction can probably be solved if the skeletal fraction (=coarse fragments with ∅>2mm) of forest soils is considered. Traditionally, the nutrient content of the coarse fraction is only considered if long term matter budgets of forest ecosystems are made.

The ionic input of the coarse fraction by weathering is regarded as negligible if actual nutritional potentials are assessed. However, recent studies showed that the coarse fraction of forest soils is more involved in biogeochemical cycles than commonly assumed (e.g. Kohler et al. 2000; Jongmans et al. 1997). The coarse fraction can store considerable amounts of exchangeable nutrient cations (Figure 7). As shown in Figure 8, nutrient adsorbing tissues such as fungal hyphae were also found in stones (Kohler 2001; van Breemen et al. 2000a and 2000b). Both facts support the hypothesis that we can not exclude a by-pass flux of nutrients via mycorrhiza directly from stones into trees (Kohler 2001). Such a by-pass flow would answer questions raised by soil chemists in Central Europe concerning observations that dramatically nutrient depleted soils (fine earths) are associated with normal tree nutrition and growth (e.g. von Wilpert et al. 2000). For this reason, the traditional practice of only considering fine earth (∅<2mm) for the assessment of a site's nutritional potential should be critically re-evaluated. For Mediterranean soils, Ugolini et al. (1996) and Martin-Garcia et al. (1999) showed that stones must not be considered as an inert matrix without any influence on ecological soil properties. According to Corti et al, (2002) rock fragments even contribute significantly to the C and N-pool of soils.

4 Conclusions

The "titration and eutrophication experiment" with forest soils in Middle- and Northern Europe yielded a drastic depletion of exchangeable basic cations. This impoverishment was mainly triggered by the introduction of stable mobile anions of strong mineral acids (sulfate, nitrate). The relation between the acidification status of soils measured by (Al+Fe)-saturation and bond strengths of Ca and Mg suggests that this depletion was self-accelerating, since bond strengths of exchangeable earth alkali ions decrease with increasing acidity. The depletion process is finished as soon as "base stocks" are reached. The widespread minimal base saturations (<5%) and especially the minimal supplies of exchangeable Mg in forest soils indicate that this state is already widely reached.

The partially extreme chemical gradients between soil aggregate surfaces and inner parts of aggregates also indicate an instationary state of forest soils. In open systems like soils, disequilibria can only be maintained by external driving forces over a longer period of time.

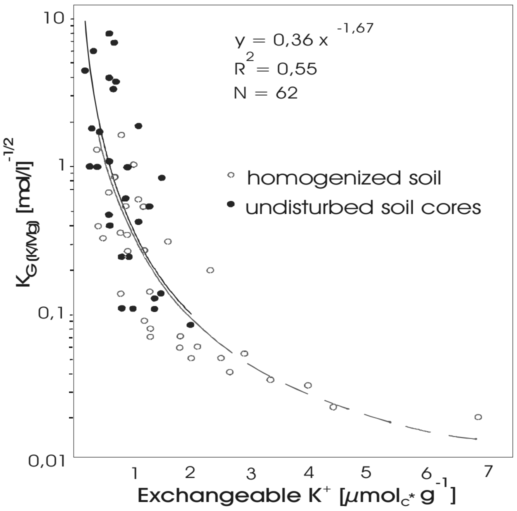

Figure 6. Relative selectivities of K binding compared to Mg calculated according to the GAPON equation plotted against the amount of exchangeable K (Ks) at different sites in South-West Germany (from Hildebrand 1994).

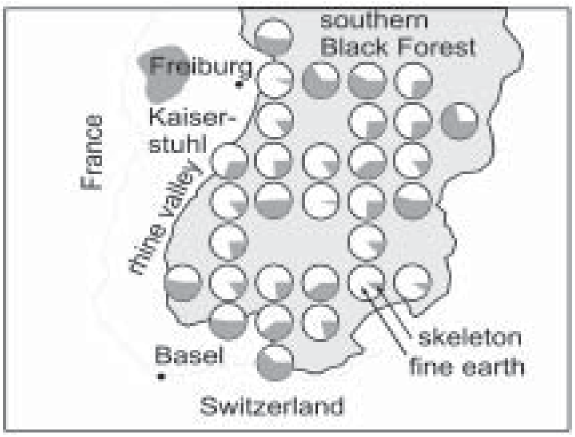

Figure 7. The contribution of the soil skeleton and fine earth to the total supply of exchangeable Mg in 0–90 cm soil depth (8x8 km sampling grid in the Southern Black Forest).

Thus, they are indicators of a drift. The chemical state of aggregate surfaces gives the direction of the drift. The slope of the gradient contains a measure of the drift velocity. In the case of potassium the K- depletion of root accessible surfaces rather than bulk supplies of exchangeable K account for widespread K-deficiencies in South Germany.

However, we strongly underestimate nutritional potentials of forest soils if we only focus on fine earth properties. The skeletal fraction contributes considerably to the pool of exchangeable basic cations. Moreover, the frequency of nutrient adsorbing tissues (fungal hyphae and/or rhizomorphae) suggests stones may be real “hot spots” of nutrient uptake.

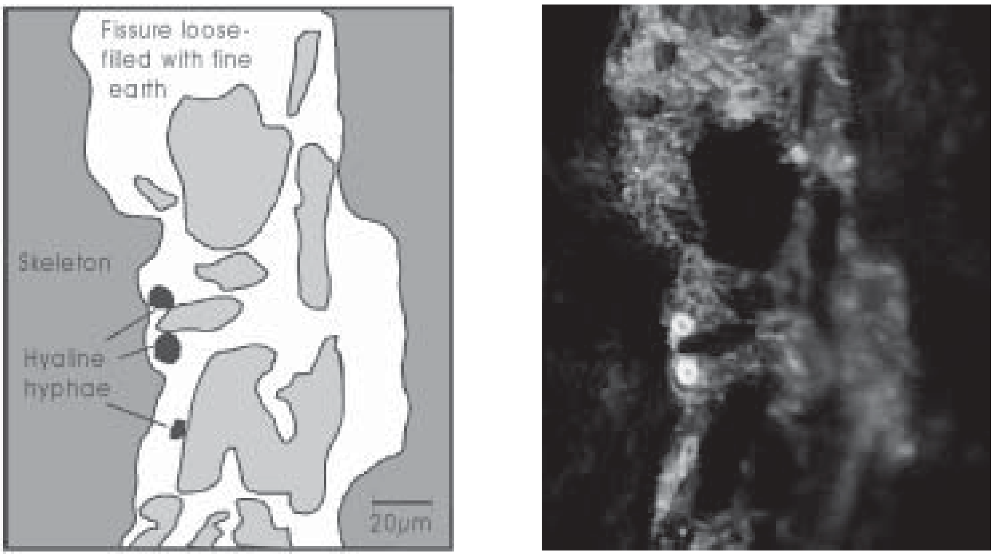

Figure 8. Fungal hyphae in a fine material filled cavity of a skeleton particle of a gneiss site. Microphoto after UV-activation (from Kohler 2001).

The "soil-deformation-experiment" in Figure 1 is "self-made" and not forced upon forestry by society in general. It is also linked with nutrient cycling since soil functioning as rooting space is drastically reduced in deformed soils (Gaertig et al. 2002).The recovery of compacted soils lasts decades. Thus, soil deformation by vehicle movement in context with site management practices has a cumulative effect. This becomes evident if we measure the CO_2-concentrations with high spatial dissolution in the top soil (5 cm soil depth). As shown in Figure 9, skid trails created network-like zones of disturbed CO_2-discharge in the top soil (foothill-zone of the Black-Forest, mature stand of *Quercus petraea*, luvisol on loess). The pattern of disturbed CO_2-discharge is widely congruent with the pattern of dead fine roots (Puls et al. 2000). These findings imply that the disturbance of gas turnover provoked by uncontrolled vehicle movement in forest sites has a widely underestimated impact on soil functioning as rooting space.

Moreover, the shift from aerobic to anaerobic metabolism in compacted soils suggests that skid trails may be unconsidered hot spots of greenhouse gas emissions such as N_2O (Butterbach-Bahl et al. 2001), thus creating a link to the greenhouse experiment.

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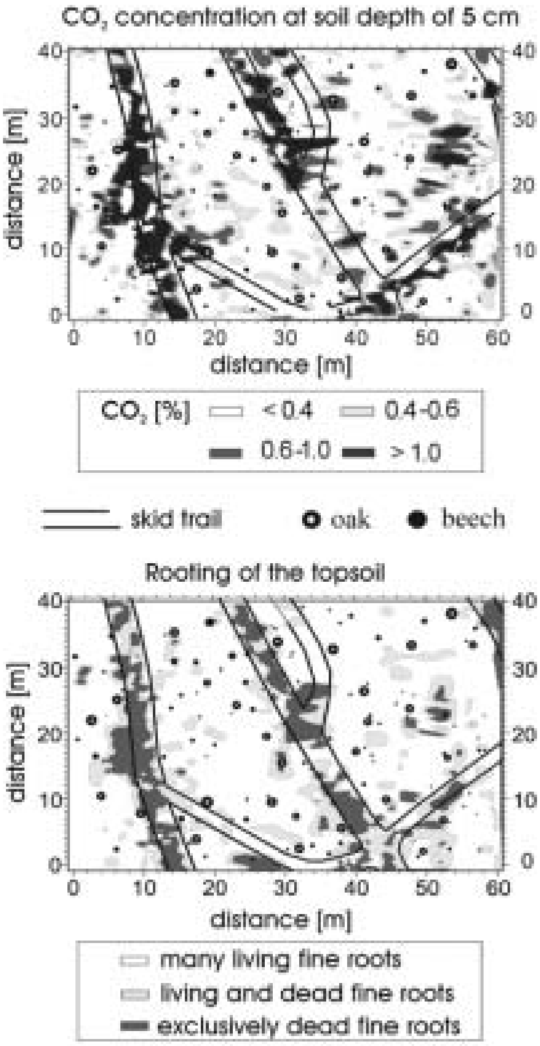

Figure 9. Distribution of CO_2 concentration at a soil depth of 5 cm (figure above) and fine root density levels (figure below) in the upper soil layer of mature stand of *Quercus petraea* in the foothill region of the Black-Forest (luvisol on loess). The positions of trees and skidding trails are also shown (from Gaertig et al. 2000).

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Investigation on the CO_2 Balance of a Scots Pine Stand Suffering from Regularly Occurring Summer Drought: Proposal of a Combination of Methods Applied in Micrometeorology and Tree Physiology

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Abstract

The Scots pine stand (*Pinus sylvestris* L.) at the long-term forest meteorological experimental site Hartheim, which is located in the southern upper Rhine plain (southwest Germany), suffers from regularly occurring summer drought. The specific site conditions, which are to be expected in Central Europe in the future, are the background for an increasing interest in the CO_2 balance of the Hartheim Scots pine stand. For its investigation, a proposal is introduced, which combines methods applied in micrometeorology and tree physiology. The CO_2 net ecosystem exchange at a reference height above the Scots pine stand will be determined by the eddy covariance approach (micrometeorology), whereas the CO_2 flux components will be analyzed by the $d^{13}C/d^{18}O$ method (tree physiology). By combining these methods, the factors governing the CO_2 exchange between the Scots pine stand and the ambient atmosphere can be assessed.

Keywords: CO_2 balance; Scots pine forest; drought; micrometeorology; tree physiology

1. Introduction

Carbon dioxide (CO_2) exchange processes at the Earth's surface are decisive for the global carbon cycle. Main sinks for atmospheric CO_2 are the dilution of CO_2 in oceans and the accumulation of CO_2 in the terrestrial biosphere. Forests represent the dominating land cover of the continents characterized by a rather long rotation period. Therefore, they play a particular

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role in the biosphere-atmosphere CO_2 exchange. Based on CO_2 exchange rates of different forest ecosystems, the carbon balance of forests can be modeled as a function of climate region, tree species, and stand characteristics (Law et al. 2000; Baldocchi and Wilson 2001).

As a consequence of climate change, weather situations causing drought will occur in Central Europe more frequent. Since the Hartheim Scots pine stand in southwest Germany suffers from regularly occurring summer drought, it is already exposed to possible climate conditions predicted by regional climate models for Central Europe (Schär et al. 2004). The current growth of the Hartheim Scots pine stand represents forest growth conditions, which can be expected in Central Europe in the future. Therefore, data on CO_2 fluxes and the CO_2 balance of this stand may help to improve the evaluation of the changing role of forests in the biosphere-atmosphere CO_2 exchange in the future or to validate CO_2 balance models for forest ecosystems.

The objective of this paper is to propose a new combination of methods applied in micrometeorology and tree physiology, which is well suited to investigate both the CO_2 balance and the sink and source strength of photosynthetic CO_2 uptake and respiration of forests. This combination of methods, which includes processes and fluxes for the CO_2 balance as indicated in Figure 1, should be tested at the Hartheim Scots pine stand.

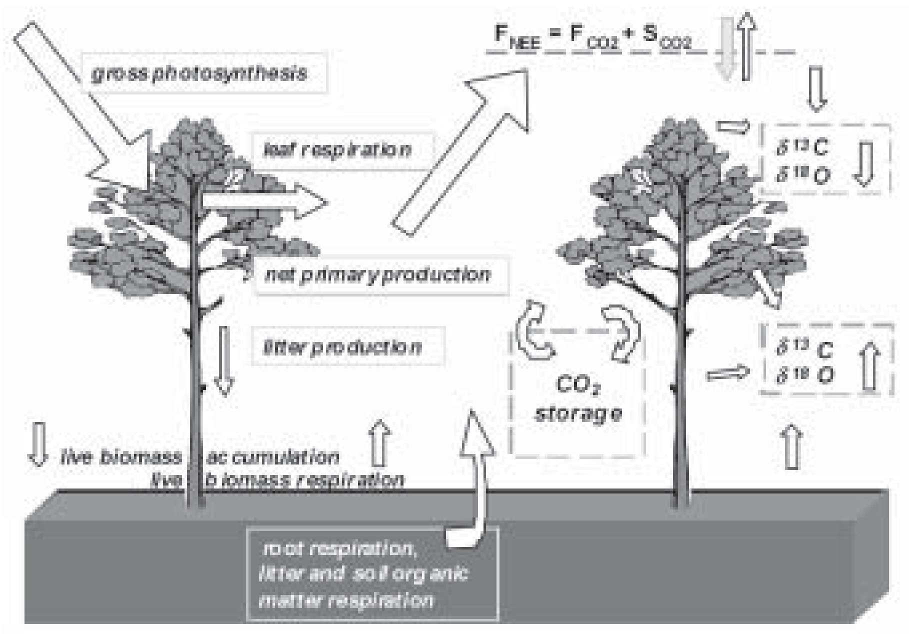

Figure 1. Processes and fluxes determining the CO_2 balance of forests; explanation of symbols in the text (after IPCC 2001).

2. Components of the CO_2 balance of a forest

Neglecting horizontal eddy flux divergence, the overall CO_2 balance of a forest may be described by (Aubinet et al. 2000):

$$\underset{\mathrm{I}}{\int_0^{h_m} S\,dz} = \underset{\mathrm{II}}{\overline{w'c'}} + \underset{\mathrm{III}}{\int_0^{h_m} \frac{\partial \bar{c}}{\partial t}\,dz} + \underset{\mathrm{IV}}{\int_0^{h_m} \bar{u}\,\frac{\partial \bar{c}}{\partial x}\,dz} + \underset{\mathrm{V}}{\int_0^{h_m} \bar{w}\,\frac{\partial \bar{c}}{\partial z}\,dz} \tag{1}$$

where term I is the CO_2 sink/source, which corresponds to the net ecosystem exchange term of carbon (F_{NEE}), term II is the turbulent CO_2 flux (F_{CO2}) at the measurement height (h_m), term III is the storage of CO_2 (S_{CO2}) below h_m, and terms IV and V represent CO_2 fluxes by horizontal and vertical advection. To provide estimates of F_{NEE}, the eddy-covariance (EC) method is used for F_{CO2}.

Detailed descriptions of the EC measurement technique, necessary corrections, data analysis and problems associated with the EC method are given in Aubinet et al. (2000), Falge et al. (2001a, 2001b, 2002a, 2002b), Law et al. (2002) and Finnigan et al. (2003).

To interpret the variability of F_{NEE} at h_m, it is essential to determine flux components of F_{NEE}. The sink and source strength of photosynthetic CO_2 uptake and respiration, respectively, are quantified by the analysis of the stable isotopes ratios $^{13}C/^{12}C$ ($\delta^{13}C$) and $^{18}O/^{16}O$ ($\delta^{18}O$) above the forest soil, in the canopy, above the canopy, and in different plant materials (wood, needles, phloem) and related to flux measurements (Geßler et al. 2001).

The isotopic values are expressed in delta notation ($\delta^{13}C$) as following Farquhar et al. (1989):

$$\delta^{13}C = \left(\frac{R_{sample}}{R_{standard}} - 1\right)1000 \tag{2}$$

where R is the molar ratio of ^{13}C and ^{12}C isotopes of the sample resp. standard (VPDB). F_{NEE} can be described by (Buchmann and Ehleringer 1998; Bowling et al. 2003):

$$F_{NEE} = F_{CO2} + \frac{dC_a}{dt} = F_R + F_A \tag{3}$$

where F_{CO2} is the total CO_2 flux density, dC_a/dt is the storage flux density, which is the time rate of change of CO_2 mole fraction (C_a) between the ground surface and measurement height h_m, F_R and F_A are the total forest ecosystem respiration and net photosynthetic assimilation, respectively. Net ecosystem exchange of $^{13}CO_2$ (F_δ) can be described according to (Bowling et al. 2001):

$$F_\delta = \left(\delta^{13}C_R\right)F_R + \left(\delta^{13}C_a - \Delta_{canopy}\right)F_A \tag{4}$$

where $\delta^{13}C_R$ is the isotope ratio of total ecosystem respiration and $\delta^{13}C_a$ is the isotope ratio of atmospheric CO_2 and (Δ_{canopy}) is the whole canopy carbon isotope discrimination by photosynthesis.

The eddy-covariance based $\delta\,^{13}C$ method (Bowling et al. 2001; Bowling et al. 2003) uses a regression between eddy-covariance measurements of CO_2 in the air and $\delta^{13}C$ to calculate the $\delta^{13}C$-flux by:

$$F_\delta = \rho \overline{w'\left[\left(\delta^{13}C_a\right)C_a\right]'} = \rho \overline{w'\left[\left(mC_a + a\right)C_a\right]'} \qquad (5)$$

where m is the slope and a is the intercept of the regression $\delta^{13}C = m\,C_a + a$.

The dependence of $\delta^{13}C$ either on photosynthetically active radiation or water availability can be differentiated by the additional measurement of $\delta^{18}O$ in plant material (Scheidegger et al. 2000). For a detailed description of the determination of $\delta^{13}C$ and $\delta^{13}C$-fluxes assessed by the EC method see Korol et al. (1999), Geßler et al. (2001) and Bowling et al. (2003).

3. Summarized results of forest-atmosphere CO_2 exchange studies

Due to the demand for long-term CO_2 exchange data between forest ecosystems and the atmosphere, regional CO_2 exchange monitoring networks in forests were established (e.g., EUROFLUX, Ameriflux) and subsumed under the global network FLUXNET in the year 1998 (Baldocchi et al. 2001). At present, the CO_2 balance of forests and CO_2 interactions between forests and the atmosphere are investigated at more than 200 tower sites. Their results are the basis for a spatially integrated analysis of the CO_2 exchange between biosphere and atmosphere.

Previous results of studies on the forest-atmosphere CO_2 exchange can be summarized as follows (e.g. Baldocchi 1997; Lindroth et al. 1998; Valentini et al. 2000; Baldocchi et al. 2000, 2001; Pilegaard et al. 2001; Falge et al. 2002a; Granier et al. 2002; Mahrt and Vickers 2002; Rannik et al. 2002; Widén 2002):

- CO_2 net ecosystem exchange (F_{NEE}) of forests depends on meso-scale climate conditions, actual weather situation, tree species, tree physiology, stand age, stand history, and silvicultural management. Since F_{NEE} is the result of the difference between gross ecosystem production (F_{GEP}) and forest respiration (F_R), F_{NEE} is sensitive to changing climate and weather conditions and shows a pronounced diurnal, seasonal, annual, and inter-annual variability.
- The mean annual air temperature and the mean air temperature over the growing season have no marked influence on seasonal and annual respiration rates. During shorter time intervals, however, there can be a significant relationship between forest respiration and air temperature.
- Under favourable site conditions, F_{NEE} is dominated by CO_2 uptake (photosynthesis) during daytime and CO_2 release (respiration) during nighttime. In deciduous forests, F_R is dominating throughout the leafless period.
- Annual F_{NEE} at EUROFLUX sites seems to be determined by F_R, which is increasing with higher latitudes in spite of a decreasing mean annual air temperature.
- With respect to seasons, most forest ecosystems are CO_2 sinks. Temperate forest ecosystems show largest annual F_{NEE} values at the southern border of their natural range. Dependent on annual weather conditions, boreal forests can either be CO_2 sources or CO_2 sinks.
- Forest respiration is strongly driven by soil and root respiration.
- Photosynthetic photon flux density, water vapour saturation deficit within the air, soil temperature, and soil moisture are driving meteorological variables for the CO_2 exchange between forests and the atmosphere.
- Drought conditions can have a distinct effect on F_{NEE}. Forests suffering from drought exhibit a reduced F_{NEE}.

4. Test site

4.1 Forest meteorological experimental site Hartheim

The forest meteorological experimental site Hartheim (47°56´04´´N, 7°36´02´´E, 201 m a.s.l.), which is run for more than 30 years by the Meteorological Institute, University of Freiburg, is located within a slow-growing even-aged Scots pine stand (*Pinus sylvestris* L.) in the southern upper Rhine plain (southwest Germany). Forest meteorological and hydrometeorological variables are recorded continuously since 1974 in order to investigate the impacts of the growth dynamics of the Scots pine stand on its radiation, heat and water balance as well as aerodynamic surface roughness (Mayer et al. 2000). The Scots pine stand extends 1 km in N-S and 0.5 km in E-W direction. It was planted in NNE-SSW orientated rows in the year 1963. The total Hartheim forest extends approximately 10 km in N-S and 1.5 km in E-W direction. Current stand density is about 800 trees ha^{-1}, current mean stand height H is 14.3 m and mean breast height diameter is 16 cm. The mean leaf area index (projected) at the experimental site was 1.6 in the year 2003. The understorey consists of various deciduous tree species (e.g., lime and beech). The soil at the experimental site is a two-layer pararendzina. The upper layer consists of sandy loam (0.15–0.8 m) and the underlying layer is alluvial gravel. Absolute mean water storage capacity of the upper layer is 80 mm, field capacity is at 31.4 vol.% and the permanent wilting point is at 11.7 vol.%.

4.2 Instrumentation and sampling techniques

The forest meteorological experimental site Hartheim is equipped with two meteorological walk-up towers (18 m and 30 m) separated by a horizontal distance of 40 m. For the long-term recording of forest meteorological variables, probes to measure wind direction, short- and long-wave radiation fluxes, net all-wave radiation, precipitation, as well as vertical profiles of dry and wet bulb temperature and wind speed are mounted at the 30 m tower (z/H = 2.1). Soil temperature is observed at 6 depths from 0.01 to 0.40 m. Soil heat flux is measured by soil heat flux plates. Volumetric soil moisture content is determined by the gravimetric method and by TDR probes (upper 0.3 m).

Since late spring 2003 a fast response infrared gas analyzer (LI-6262; LI-COR, USA) and a sonic anemometer (R2; Gill, UK) are installed at the top of the 18 m tower (z/H = 1.3) to determine both the CO_2 and water vapor fluxes above the Scots pine stand using the EC method. Photosynthetic photon flux density is measured at different levels above and within the Scots pine stand.

Data obtained from the ‘slow’ instrumentation at the 30 m tower are recorded (CR23X; Campbell, USA) as 10 min mean values and 10 min sums, respectively. Data obtained from the LI-6262 (10 Hz) and the R2 (20.8 Hz) are stored as 30 min files.

Analyses of the stable isotopes ratios $^{13}C/^{12}C$ ($\delta^{13}C$) and $^{18}O/^{16}O$ ($\delta^{18}O$) above the forest soil, in the canopy, above the canopy and in different plant materials (wood, needles, phloem) are projected to quantify the sink and source strength of photosynthetic CO_2 uptake and respiration, respectively. To determine $\delta^{13}C$ of atmospheric CO_2, ambient air will be collected with vacutainers at the forest meteorological experimental site Hartheim and analysed with a gas chromatograph coupled to an isotope ratio mass spectrometer (Delta Plus; Finnigan MAT GmbH, Bremen, Germany). Conditioned (oven-dried and homogenised) phloem samples will be analysed with an elemental analyser (NA 2500; CE Instruments, Milan, Italy) for $\delta^{13}C$ analysis and with a high temperature conversion / elemental analyser (TC/EA Finnigan MAT

GmbH, Bremen, Germany) for $\delta^{18}O$ analysis, both coupled to an isotope ratio mass spectrometer (Delta Plus; Finnigan MAT GmbH, Bremen, Germany) by a Conflo II interface (Finnigan MAT GmbH, Bremen Germany).

5. CO_2 flux above the Hartheim Scots pine forest in the extremely dry August 2003

A record-breaking heatwave affected the European continent in summer 2003 (Schär et al. 2004) causing extreme drought stress in forests. The impacts of this summer drought on F_{CO2} during the growing season are exemplarily illustrated for August 2003 (Fig. 2)

Compared to September 2003, the mean diurnal cycle of F_{CO2} in the dry August 2003 exhibits the following characteristics:

- Mean CO_2 uptake (negative F_{CO2}) in the morning (around 8 CET) occurred a little earlier in August than in September.
- The highest mean CO_2 uptake was clearly lower in August (-2.6 µmol m^{-2} s^{-1}) than in September (-8.2 µmol m^{-2} s^{-1}). It was observed around 10 CET, i.e., two hours earlier than in September.
- Mean CO_2 release (positive F_{CO2}) in the afternoon was recorded at 15 CET in August 2003 and between 18 and 19 CET in September.
- Based only on F_{CO2}, the Hartheim Scots pine stand represented a CO_2 source in August 2003, but a 1.4 times higher CO_2 sink in September 2003.

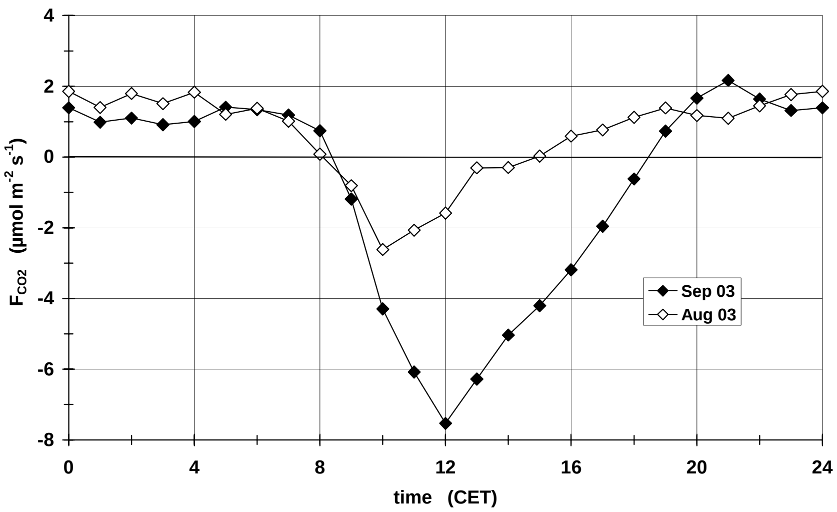

Figure 2. Mean diurnal cycles of the turbulent CO_2 flux (F_{CO2}) of the Hartheim Scots pine stand (*Pinus sylvestris* L.) in August and September 2003 (relative measuring height z/H = 1.3).

6. Perspective

The Scots pine stand at the forest meteorological experimental site Hartheim suffers from regularly occurring summer drought, which represents a future climate scenario predicted by regional climate models (IPCC 2001) and, therefore, reflects possible future growth conditions for forests in Central Europe. In spite of the extensive observational network on CO_2 exchange between forests and the atmosphere (FLUXNET), further investigations on the CO_2 balance of forests, particularly under extreme site conditions such as drought, are necessary to improve the current understanding of the CO_2 exchange between forests and the atmosphere due to climate change (Valentini and Baldocchi 2002). A great portion of year-to-year variability of F_{NEE} at forest sites can be attributed to extreme site conditions, e.g. during the growing season, reduced water availability leads to a lower carbon uptake, whereas respiration is increased by elevated air temperature (Granier et al. 2000; Law et al. 2000; Meyers 2001; Falge et al. 2002b).

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Forestry Management Options for Water Preservation

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Abstract

Forested areas have a high potential for water preservation and are reserves for low polluted water. The long-term survey of groundwater quality shows that even below forest floors water quality drops. Nitrate concentrations increased from the typical output levels of forest ecosystems of 2 to 3 mg/l up to 5 to 6 mg/l and more within the last decades. At the landscape level, a prognosis of nitrate concentrations in the groundwater succeeded by using site parameters (critical load exceedance for N, C-stock in the mineral soil (0–30 cm), annual precipitation, C/N gradient between humus layer and the upper mineral soil (0–5 cm), Ca-stock in 0–5 cm and pH (KCl) in 10–30 cm) from the statewide Soil Chemical Survey in a multiple linear regression model. This model explained about 60% of the variance in nitrate concentrations in the groundwater. Thus the first step towards closing the gap between locally related process studies and landscape related practical planning is done.

Keywords: Nitrogen saturation; soil acidification; liming; forest management; water quality

1. Introduction

Forests in Germany and especially in the federal state of Baden-Wuerttemberg, where the results of this study were gained, cover about one third of the landscape surface in non-urban areas. It is well-known that water quality of ground water and streams is better in forested areas than in the open landscape. Especially the contamination with nitrate and pesticides is lower in forests, whereas reaction products of soil acidification like e.g. dissolved organic carbon (DOC) and aluminum are an increasing problem in forest seepage water under the influence of anthropogenic depositions which are disturbing the element budgets and the functionality of forest ecosystems in large areas of Central Europe (v.Wilpert 2002; v.Wilpert and Zirlewagen 2001). Gäth and Frede (1991) e.g. showed that nitrate concentrations depend

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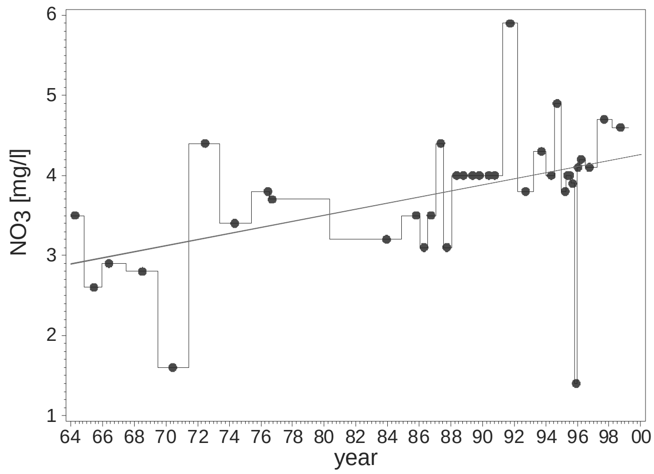

Figure 1. Nitrate time series of a spring at granite bedrock in the Southern Black Forest with a catchment, totally covered by forest (Data from the groundwater survey of the "Landesanstalt für Umweltschutz Karlsruhe", LFU).

on the proportion of forested area in catchments and are lowest in forests. Nevertheless, also the nitrate load increases in forested catchments because of nitrogen deposition, which causes subsequently increasing nitrogen saturation in forest ecosystems. A typical example for the development of nitrate concentrations in the groundwater over time is given for a spring at granite bedrock in Todtnau in the Southern Black Forest (Figure 1).

In this example the nitrate concentration increased from the very low values of between 2–4 $mg_* l^{-1}$ in the 1960s to about double the value in the 1990s. In areas with Triassic Sandstones this increase was even higher.

In the state of Baden-Wuerttemberg 70% of the drinking water originates from small, local sources (Krenzke 2000). Water resources mainly feed these from the upper ground water or surface water, which is collected in drinking water dams. Surface-near water resources is likely to be influenced by environmental changes like anthropogenic acid and nitrogen deposition from the air. In this context, the potential of forest management to counteract these unfavorable industrial hazards is of high interest. Moreover, the preservation of the high water quality in forested catchments is important, because the water from the forests is often needed for mixing with water of lower quality originating from agricultural areas.

The aim to observe the development of water resources in the forest covered proportion of the landscape, in order to identify forestry management options for sustainable water preservation and to develop strategies for counteracting tendencies which lead to deterioration of water quality. This is also addressed the recently created water directive which provides a guideline for water preservation policy of the European Union (European Parliament 2000). It prescribes the observation of "water regions" and the development "of counter actions against processes which lead to deterioration of water quality or quantity".

The aim of this study is, on the one hand, to identify causal relationships between influencing environmental factors and processes acting at the soil surface and water quality. On the other hand, different forestry management options, like e.g. tree species selection, avoiding clear-cuts or forest liming, will be discussed in terms of their potential for long-term water preservation.

Firstly, the causal relationship between surface processes and water quality are identified by multiple linear regression analyses for the example of predicting nitrate concentration in the ground water using weather data, nitrogen deposition and soil data as predictor variables. Then single examples of the influence of forest management characteristics are discussed in a more qualitative way, using the results from different ecosystem case studies in the region.

2. Relation between surface processes and nitrate concentrations in the ground water

The Office of Environment Preservation ("Landesanstalt für Umweltschutz") in Karlsruhe started a groundwater survey network in the early 1960s, with 388 measuring points in the Black Forest. The database contained nitrate concentrations, which were measured at varying dates, besides a detailed set of describing data. The mean nitrate concentrations in the ground water of the 1990s were used as target variable for regression analyses. The sampling points of the ground water network are accidentally distributed according to the location of springs and wells. In the same area of the Black Forest, the state-wide Chemical Soil Survey has106 grid points in a 8x8km grid. The dataset of the Chemical Soil Survey contains descriptive data like soil type, humus form or soil texture and analytical data. Critical loads for sulfur and nitrogen as well as critical load exceedances were calculated using the SMB model (Sverdrup et al. 1990) for each grid point. Depositions were assessed from EMEP data according to Gauger et al. (1995).

The first step of evaluation was to assign grid points of the soil chemical survey to the locations of the groundwater network. This assignment was done according to the following rules:

- No distance larger than 4 km was accepted between the points of the groundwater and the soil survey;
- The corresponding sampling points should be at the same geologic unit:
- Sampling points of the soil survey were only included if they were situated orografically above the corresponding spring.

Thus at least one sampling point of the groundwater network could be assigned for each 55 sampling points of the Chemical Soil Survey. Since the datasets were not always complete only between 30 and 40 datasets were ready for multiple regression analysis at the end.

In a first explorative phase, potential predictors were examined in univariate scatter plots (Figure 2). There variables describing weather and nitrogen deposition, the N-stock in the soil, humus and C-stock and the more variable chemical properties were examined.

The scatter plots in Figure 2 show no strong regressions. Only height a.s.l. and annual precipitation displayed coefficients of determination >0.2–0.3 but they were highly correlated among each other. Thus height a.s.l. was excluded from further analysis.

A tendency towards positive correlations to the target variable (nitrate concentration in the groundwater) displayed the mean annual temperature, the mean critical load exceedance of the years 1993–1995 and pH values. All other potential predictors were negatively correlated to nitrate concentration. Unexpectedly, the characteristics of the N-stock in the humus layer (total N content and C/N ratio) showed no dependency to the nitrate concentrations in the groundwater whereas the N-stock in the deeper mineral soil and the relation between the C/N ratios of the humus layer and the upper Ah horizon (C/N-gradient) showed a tendency to negative correlations indicating nitrate concentrations to decrease when nitrogen is stably bound to the mineral humus. The same tendency showed the C-content.

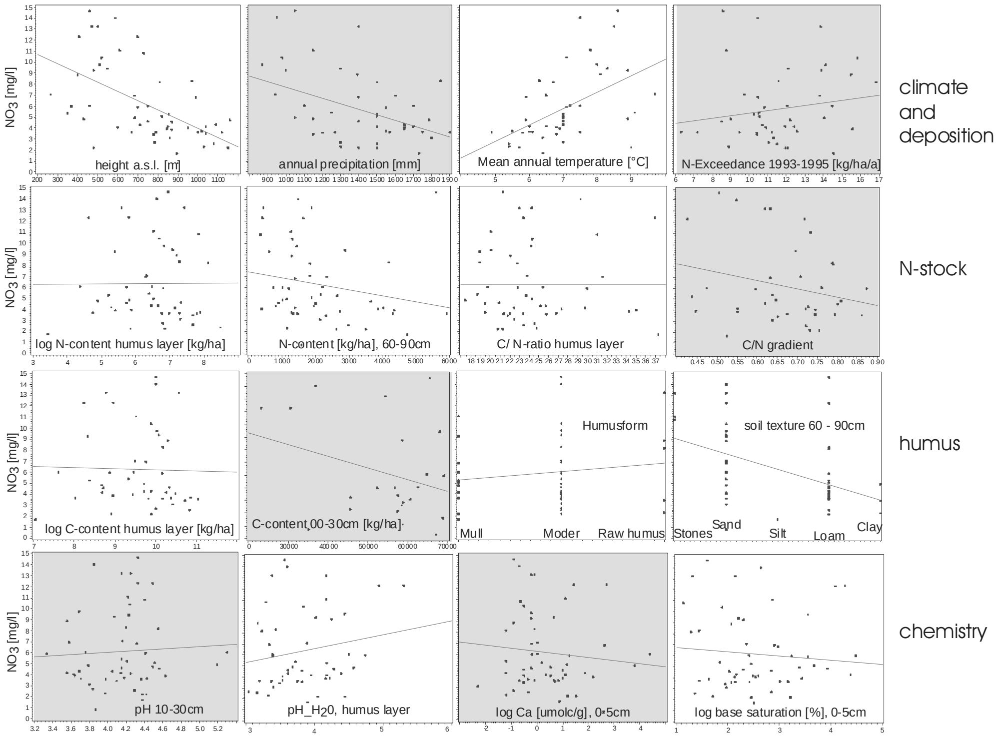

Figure 2. Scatter plots for potential predictor variables of nitrate concentrations in the groundwater. Grey shaded fields are the variables identified in multiple regression analysis as significant predictors. Groundwater data from LFU Karlsruhe.

Nitrate concentrations in the groundwater were predicted using linear regression analyses by the REG procedure of the SAS statistical package (SAS Institute 1990). In order to find the model providing "best" prediction of chemical soil attributes, stepwise regressions with forward-selection technique were performed using the default significance level of 0.5 for parameter entry. The maximum number of independent variables in the models was limited to 6 parameters.

The following model could be identified by the multiple linear regression analyses, where the critical load exceedance for N, C-stock in the mineral soil (0–30cm), annual precipitation, the C/N gradient between humus layer and the uppermost mineral soil (0–5cm), Ca stock in 0-5cm and pH (KCl) in 10–30cm were identified as significant predictors (Figure 3).

The studied residuals were equally distributed above the predicted values and stayed within the tolerable range of between ± 3 with residuals not larger than ±2. The R^2 value of 0.62 (adjusted R^2 = 0.52) is high with respect to the low number of 37 complete datasets, which could be included into the regression analyses giving 24 degrees of freedom.

Figure 4 presents the spatial distribution of the relative residuals (observed-predicted / observed) in the Black Forest.

Most of the relative residuals amounted to not higher than the factor of ±0.5–0.7 of the observed values. Only few outliers with maximum relative residuals up to 1.8 lie mainly in the region of the eastern frontier of the Black Forest where nitrogen depositions are at minimum and therefore the model tended to over-estimation.

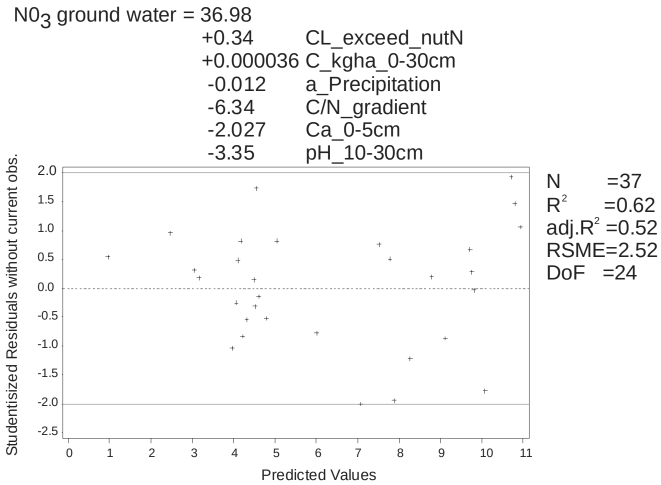

Figure 3. Parameters and statistical characteristics of the model for prediction of nitrate concentrations in the groundwater using soil- and environmental predictors. Distribution of studentisized residuals in dependency of the predicted values.

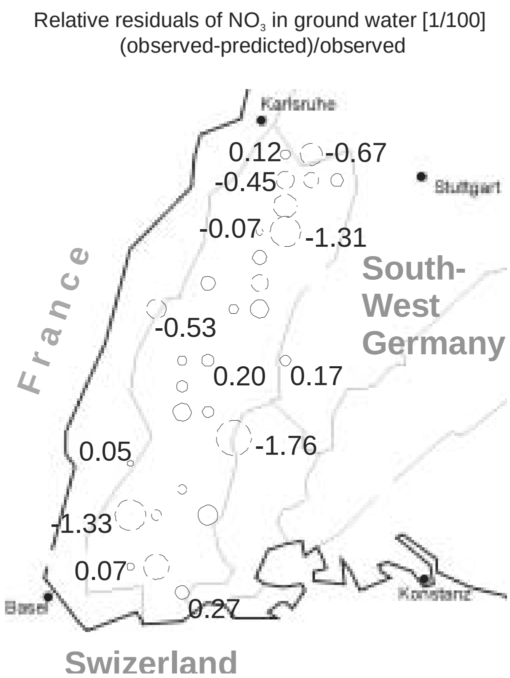

Figure 4. Relative residuals between observed and predicted nitrate concentrations in the ground water in the region of the Black Forest (South-West Germany). Closed circles indicate observed values to be higher than predicted, broken circles opposite. (Groundwater data from LFU Karlsruhe)

3. Identification of the potential of forest management for water preservation

In this chapter, examples of aspects of forest management will be presented which are likely to provide a certain potential for water preservation and therefore could serve as key characteristics for defining a long-term water preservation strategy in forested catchments. The results were gained in several ecosystem case studies in Southwest Germany.

3.1 Tree species composition of stands and nitrogen budget

Tree species composition influences decisively the flux of nitrogen compounds through forest ecosystems. The main variable is the deposition, which is about 30–50% lower in broad-leaved stands than in coniferous stands (v. Wilpert et al. 2000).

The total nitrogen balance (Figure 5) was characterized in the Convent Forest ecosystem study (South-West Germany, near Freiburg) by flow equilibrium on a high level for both spruce stands. About 60% of the deposited nitrogen was washed out at 180 cm depth. In contrast, only 25–30% (about 4 kg ha^{-1} a^{-1}) of the deposited nitrogen left the ecosystem in beech dominated stands, which were directly neighbored to them.

Spruce stands were a net nitrate source, whereas the beech dominated stands were a net nitrate sink. Ammonium was either completely assimilated or nitrified in the soils of all stands.

3.2 Forest liming

Forest liming is thought in practice to be an effective countermeasure against soil acidification. Even if liming provokes unintended side effects like an overshooting nitrification leading to nitrate mobilization for few years, the long-term goal of forest liming is to enhance the biological activity of forest soils, which was damaged by acidification. The stable storage capacity of forest soils for nitrogen and carbon will indeed be enhanced as a consequence of a more favorable chemical milieu, as Hildebrand (1996) showed by use of the results of the soil chemical survey. This is the aim to be achieved by forest liming. Comparable results presented Schäffer et al. (2001) from long-term liming trials in Baden-Wuerttemberg where nitrogen- and carbon stocks were elevated at the limed plots as well as fine root density. Thus catchment liming should also lead to more closed element cycles and therefore to better water quality.

In the area of the drinking water dam “Kleine Kinzig” two partial catchments were examined which were limed with different intensity. The area of the catchment of “Huttenbächle” was nearly completely limed, only at the northern parts, far away from the stream, 20% of the catchment area were not limed (Figure 6).

In contrast, the area of the “Teufelsbächle” was only to about 55% limed. There liming took place after 1985 which makes the “ecosystem answer” plausible to be not yet fully developed. In Figure 7 initial ecosystem reactions are presented in the time series of stream-chemistry parameters.

At the beginning of the time series of stream chemistry, the liming campaign was performed in both cases. At that time the pH values as well as Calcium concentrations were very comparable between the two catchments. Subsequently the difference between the catchments increased for both parameters by the time. This result can be interpreted as the initial effect of the additional buffering capacity which was provided by liming and which slowed down the acidification trend driven by acid depositions. Other reactions of the ecosystem upon liming, which are slower developing, like the enhancement of stable nitrogen immobilization in the mineral humus and subsequently a decrease of nitrate concentrations in the stream could not yet be observed in this case. The nitrate concentrations stayed at a comparable level in both catchments.

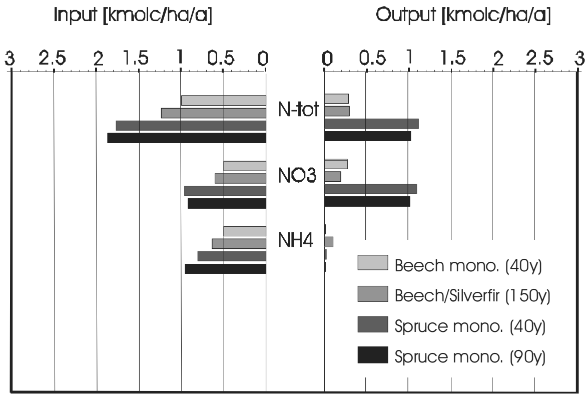

Figure 5. Comparison of nitrogen balances of deposition input above the crown layer and nitrogen output in 180cm soil depth in a beech mono-cultured, a beech/silver fir mixed and two mono-cultured spruce stands (v. Wilpert et al. 2000).

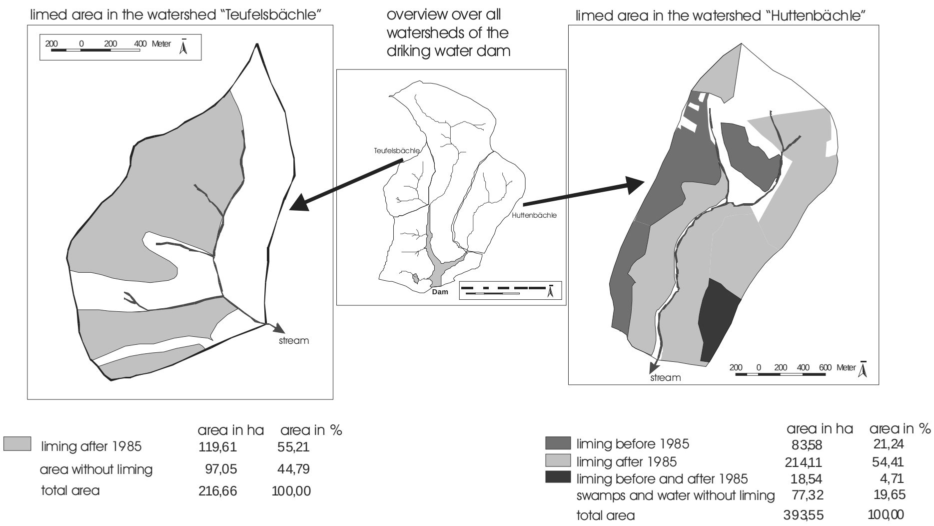

Figure 6. Differently limed catchments of the catchment of the drinking water dam "Kleine Kinzig", Middle Black Forest, Southwest Germany.

4. Conclusion – demand for further research

Different ecosystem case studies demonstrate that there are indeed substantial abilities for forest management to influence element budgets of forest soils towards stabilization, even in times of anthropogenic acid and nitrogen depositions. As the main influencing variables, which can be used in models for predicting the potential of forest management for water preservation, stand composition (broad-leaved vs. coniferous trees), regeneration technique (clear-cut vs. gap oriented), and the existence of pre-regeneration have been identified (v. Wilpert 2002; Zirlewagen and v. Wilpert 2001; v. Wilpert et al. 2000).

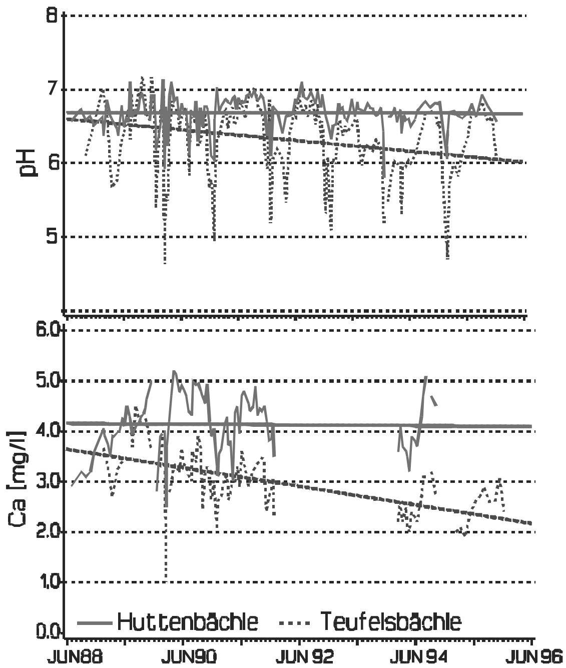

Figure 7. Time series of pH-value (above) and Ca concentrations (below) in the stream water of the partial catchments Huttenbächele (solid lines) and Teufelsbächle (dotted lines).

In a multiple linear regression model it succeeded to predict the mean nitrate concentration in the ground water with a certainty of about 60%, using deposition-, weather- and soil variables as predictors. This result does surely not exhaust all possibilities of this approach because of the comparably low number of replications. In this sense this study is to judge as a pilot study. At present it can be stated that there do exist interactions between soil surface processes and the water quality in groundwater or streams.

In the catchment study "Kleine Kinzig", the effect of catchment liming on stream water quality was examined under the specific influence of comparably high deposition load in nitrogen and acidity, which was typical for Central Europe during the last decades. In this case study the element outputs with the stream water of a 80% limed and a 55% limed catchment were compared. There was a clear tendency towards higher pH-values, and Ca-concentrations in the stream at the more limed catchment.

The transfer of the process knowledge, which was generated in several local case studies to the landscape scale, was performed very successfully by generating a regionalization method which allowed to predict soil chemical properties by using stepwise multiple regression analyses in a model area of 225 km² in a spatial resolution of 50x50m. As key predictors, topographic site attributes were integrated into the regression models by means of digital terrain analysis using a digital elevation model with a grid size of 50 m. The statistical security of this prediction was between 50 and 70% (average 60%) explained variance (Zirlewagen 2003; Zirlewagen and v. Wilpert 2001, 2004). These new regionalization schemes combined with the broad pool of data from the different networks of environmental monitoring (e.g. the chemical soil survey with its 8x8km grid) provide a sound basis for statewide landscape planning. Especially the gap between the local process level and the near landscape level (e.g. small catchments below 100 ha) has to be closed by scale typical characteristics according to the "scaleway" concept of Roth et al. (1999). That is the main task for further research in this field.

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Cycle of Water and Biogenous Elements in the Forest Ecosystems in Latvia

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Abstract

The long-term forest hydrological field data obtained over the period of 1963–2003 from three catchments with peat soils and two catchments with hydromorphic mineral soils have been supplemented with biogenous elements budget data collected since 1997. A forest drainage system has been used for a field experiment to study the hydrological parameters associated with the forest stand structure and meteorological conditions, and to estimate the influence of drainage on water quality. The hydrological regime of drained forests on deep peat soils and those on hydromorphic mineral soils differs significantly. The precipitation inputs of N-NH_4^+, N-NO_3^-, P-PO_4^{3-} and K^+ into the forest ecosystem exceed their outputs, whereas the losses of Ca^{2+} and Mg^{2+} from the forest via ditch runoff exceed their inflow by precipitation water. Annually, the amount of N-NH_4^+, N-NO_3^-, P-PO_4^{3-} that has reached the soil via precipitation, is higher in an open area, while the amount of K^+, Ca^{2+} and Mg^{2+} is higher in the precipitation water that has flown through the forest canopy.

Keywords: drained forests; biogenous elements; forest hydrology.

1. Introduction

Forest hydrology and biogeochemical cycling of a forest ecosystem has been previously examined using different approaches: catchment (Likens et al. 1977; Moldan and Cerny 1994), forest stand (lysimeter) method (Ulrich et al. 1979; Matzner 1988), patch scale roof experiments (Bredemeier et al. 1998), chronosequence studies (Schaaf 2000; Dambrine and Ranger 2000), applied tracer studies (Kendall and McDonnel 1998), combined model calculations (Item H 1974; Federer and Lash 1978; Johnson and Lindberg 1992; Liu et al. 1992).

The catchments of drained forests in Latvia are suitable objects for the hydrological research and study of the cycle of biogenous elements, as far as a quantitative evaluation of the water balance elements is concerned, is considerably easier in the small catchments. A net of drainage ditches allows to determine precisely the amount of runoff; the confined aquifer water discharge, which is characteristic to the swampy territories of Latvia, prevents from the precipitation water infiltration through the soil to deeper layers. This is characteristic to the dry areas and difficult to quantify – here, it, after certain retention in the peat, together with the discharging confined aquifer water and inflowing water from adjoining dry areas, comes just to the ditches (Zalitis 1983).

In the climatic conditions of Latvia, rainy years, caused by cyclonic activity, alternate with dry years. As shown by Pastors (1972), the long-term average water balance for the country's land area, calculated by the Riga Hydrometeorological Observatory, is as follows:

755 mm (precipitation) = 238 mm (runoff) + 517 mm (evaporation) (1)

Thus, on average, 32% of the total sum of precipitation leaves via rivers, and 68% evaporates. However, the average long-term values for different sites are very variable. The amount of precipitation varies between 600 and 970 mm, runoff between 130 and 380 mm, and evaporation between 440 and 650 mm. The annual precipitation over Latvia in 1891 to 1965 varied between 348 mm (1939) and 1007 mm (1928). The influence of territorial differences on the total dispersion of precipitation is significant, although relatively small (28%). 72% of the variations of quantity depend on meteorological conditions (Anonymous 1968).

The paludification of forests, a process that can take several thousand years, cannot be explained by variability in meteorological conditions alone. The causes are also linked to the geological and hydrogeological peculiarities of the regions. The location of wetland forests in Latvia reveals that their distribution has a logical pattern. Surprisingly, in those areas where the average annual precipitation is higher, there are fewer wetland forests. 86% of waterlogged forests with peaty soils are found in areas of confined aquifer water discharge. This fact is important when analyzing both the water balance and the cycle of biogenous elements, especially Ca^{2+} and Mg^{2+}, in drained forests (Zalitis 1983).

According to the hydrological conditions in Latvia, the water balance equation for the waterlogged forests is as follows:

$$N + P_p + P_s = Q + ET \pm \Delta W \qquad (2)$$

where: N is precipitation, a permanently positive value; P_p the confined aquifer water discharge (in the woodlands with deep peat soils this is a positive value, and in the woodlands with hydromorphic mineral soils a negative value close to zero); P_s is the water inflow from the surrounding dry areas, a permanently positive value; Q is runoff, a permanently positive value; ET is evapotranspiration, a permanently positive value and ΔW is the fluctuation of the water capacity in the active layer of soil, in the growing season a permanently negative value (Zalitis 1983; 1997).

The aim of this paper is to present the water balance equations used for describing the water cycle in forests in the conditions existing in Latvia. The cycle of the most important biogenous elements is investigated by using the catchment approach.

2. Methods

In order to assess the influence of silvicultural activities (especially hydrotechnical drainage) both on forest stand structure and hydrological conditions, the Vesetnieki Station of

Permanent Ecological Research was established in 1960 – at the same time as the area was being drained. The woodlands, representing a grass/transitional bog, were drained by laying out 1.1 to 1.2 m deep ditches spaced about 180 m apart, with buried drains 0.8–0.9 m deep laid two years later on an ad hoc basis.

Forty years after drainage, the area now carries mainly mixed conifer and broadleaf stands with stand composition 6P3S1B (i.e. on average 6 pines (*Pinus sylvestris* L.), 3 spruces (*Picea abies* (L.) Karst.) and 1 birch (*Betula pendula* Roth.) tree per 10 trees. In 1960, the forest standing volume was on average 40 m^3 ha^{-1}, in recent observations it was noted that it has increased to 292 m^3 ha^{-1} (Indriksons and Zalitis 2000).

The following measurements of water balance elements have been carried out since the drainage took place (see also Figure 1):

(a) precipitation in an open place with 0.2 mm precision (*Tretiakov* precipitation gauge), read daily;
(b) net precipitation falling through the forest canopy, from 180 rain collectors on 10 line transects in different forest stands with 0.2 mm precision, read once every 10 days over the growing season (May 1 to October 31);
(c) density and thickness (with 1 cm precision) of the snow layer in 10 different forest stands, read once every 10 days, and during the snowmelt, once every 3 days;
(d) water level in the river Veseta with 1 cm precision – daily;
(e) the groundwater level in 220 observation wells with 1 cm precision, read once every 5 days;
(f) ditch runoff in m^3 with the runoff recording device (*Valday* type) changed once a week (in 5 hydrometric posts).

To describe the redistribution of precipitation in the forest stands the interception has been calculated. Interception I is given by:

$$I = N - N' \tag{3}$$

where N is the precipitation in an open place [mm]; N' is the throughfall [mm].

To describe interception and formalize the results, all transects have been divided into two groups. In the first group, transects were established in the young stands, or in the stands where growth is radically influenced by the drainage. The second group contains stands in originally dry forest sites and also in old stands in drained forests. The places of location of the rain collectors were selected using the method of partial randomization, i.e. disposing them in a transect according to the chance principle. The rain collector consists of a glass bottle and a plastic funnel with collecting surface of 55 cm^2. The bottle is put into a pit made with cylindrical spade, so that the above surface of the funnel is at the same level as the soil surface.

Interception measurements in the growing season are formalized as an equation of linear regression of the function:

$$I = f\,(N, n, T) \tag{4}$$

where N is the throughfall [mm]; n is the number of rainy days, in which $N \geq 1.0$ mm and T is the time after drainage [years].

Since the stemflow in stands of conifers is considered as generally minor (Mitscherlich 1981; Zalitis 1983; Wohlrab et al. 1992; Dubé et al. 1995), it was not included in balance.

The discharge of confined aquifer water P_p is the most significant component in the water balance of waterlogged forests with peat soils. In contrast, the values of these components in the originally dry forests are usually negative and barely measurable. Following Darcy's law:

$$P_p = f\,(\Delta h, k, L, H) \tag{8}$$

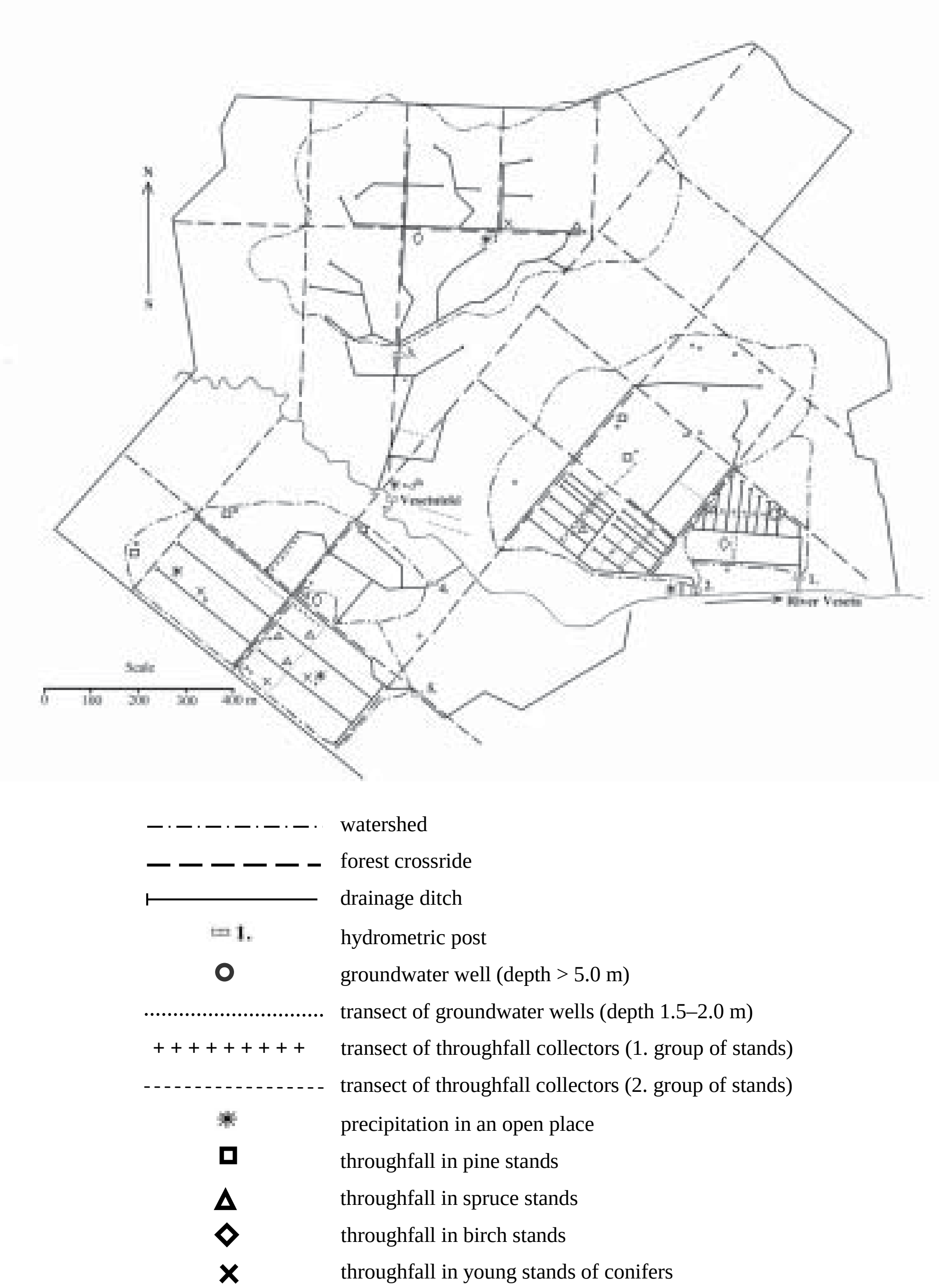

Figure 1. Scheme of the monitoring sites and distribution of collectors in Vesetnieki Station.

where Δh is the water pressure (the difference between piezometric surfaces of groundwater in the confined aquifer (h) and in the unconfined aquifer (H)); k is the coefficient of percolation and L is the thickness of the ground layer (length of the percolation route).

The calculation of P_s values is possible by using a modification of Darcy's equation:

$$P_s = B \cdot H \cdot k \cdot I \tag{9}$$

where *B* is the length of the contour of surrounding dry areas; *H* is the depth of water layer; *k* is the coefficient of percolation and *i* the slope of water flow.

In addition, measurements of the biogenous elements balance have been carried out twice a month since April 1997. The quality of precipitation water reaching the forest ecosystem has been analyzed using water from 24 precipitation collectors (Figure 1). The size of each collecting surface is 106 cm^2, and the collectors were disposed as follows: 5 in an open area, including 1 at the station's centre, 4 around the perimeter of the experimental area at a distance of 1.0 km from the station's centre; 19 in the forest stands of pine (5), spruce (6) and birch (4), as well as in a young stands of conifers (4). To calculate the output of nutrients in streamflow, one-litre samples were taken at each of 5 hydrometric posts. Three of these samples are of ditch runoff water from the catchments with a deep layer of peat, while the remaining 2 hydrometric stations monitor catchments with hydromorphic mineral soils. For each water sample the concentrations of $N–NH_4^+$, $N–NO_3^-$, $P–PO_4^{3-}$, K^+, Ca^{2+}, Mg^{2+} and pH reaction were determined at the Forest Soil Laboratory of the Latvian Forestry Research Institute "Silava". The number of water sample analyses made till July 2001 totals 17 600 (49 replications). The concentration of nitrogen, phosphorus and potassium was determined by the photometric method, that of calcium and magnesium – by titration by Trilon B.

Using the data on the concentration of biogenous elements in the ditch runoff water derived in the Vesetnieki Station as well as the data read by the continuous runoff volume recorder, the models were worked out in the form of a regression equation, which characterize the runoff of biogenous elements from the Latvian waterlogged forest ecosystems. Having commenced the development of models, it was taken into consideration that the runoff amount is different in each catchment, and significant differences in the water runoff amount exist also between seasons. The largest runoff is stated in spring and the lowest – in summer.

To extrapolate the concentration of biogenous elements in the ditch runoff during the period between the days of sampling, an attempt was made to correlate the concentration of biogenous elements in the water with the water runoff volume (mm $daily^{-1}$) on the day of sampling. A relation, in the form of a linear regression, was calculated for each element analyzed, each of the five catchments, for each of the four seasons.

For the calculation of the biogenous elements outflow on the water sampling day, the concentration on the relevant day was multiplied by the runoff amount on that day. To determine the concentration of biogenous elements for the days between water sampling times for the seasons when a statistically significant relation between the concentration of elements and runoff of the relevant day was stated, it was calculated after the derived equation. In the seasons when the significant relation between the runoff amount and concentration of elements was not stated, the average concentration of elements determined in the relevant season is used for the calculation of the runoff of those elements.

In that way, the concentrations of biogenous elements in the ditch runoff water were calculated or assumed for each day for each catchment. Multiplying the relevant daily water runoff amount by the concentration of biogenous elements and dividing into the drained area of catchment, the runoff of elements was derived in kilograms from a hectare daily. Summing up the calculated daily value of the runoff of biogenous elements, we derived the annual runoff from the relevant catchment.

In the present work, the inflow of biogenous elements, for each of 5 catchments, is estimated on a fortnight basis as the weighted arithmetic means in line with the forest structure of each catchment.

3. Results and discussion

Precipitation

The annual precipitation from continuous observations made at Vesetnieki station from 1967 till 2000 has varied between 499 mm (1975) and 1016 mm (1980). During the growing season (May to October) the precipitation has varied between 169 mm and 695 mm correspondingly. The mean value of precipitation in the growing season is 448 mm, which is close (95%) to the long-term mean value for Latvia – 470 mm. Over the last three years, during which the estimates of the biogenous elements balance were made, the mean value of precipitation was somewhat less at 434 mm and was obviously influenced by the dry (329 mm) summer of 1999.

Interception

Post-drainage changes in the forest stand structure have transformed the redistribution of precipitation in the forest ecosystems to a great extent.

Over the observation period at Vesetnieki the forest stands have aged by more than 30 years. After the drainage, the increase in interception associated with an incremental increase in standing volume is an important factor in establishing favorable soil moisture conditions under the forest for optimum growth. At the same time, as the tree height increases, the influence of wind on interception intensifies, reducing the amount of water bounded by canopies. For the first group:

$$I_1 = 0.17N + 0.78n + 2.0T - 31 \text{ (mm, } R^2 = 0.79) \qquad (5)$$

For the second group:

$$I_2 = 0.2N + 0.52n - 0.4T + 16 \text{ (mm, } R^2 = 0.73) \qquad (6)$$

The limits are: $170 < N < 700$; $30 < n < 80$; $8 < T < 35$.

The interception in the first group has increased by about 2 mm a year, but in the second group it has decreased a little during the time. The analysis performed confirms that interception in the drained forests is obviously about 60 mm higher than in the undrained forests with low productivity and crop density less than 0.7.

For the time being we cannot complete the water balance for separate forest stands. This is only possible at the catchment scale, within which the structural diversity of stands is frequently high. At Vesetnieki, there are five such catchments, and each of them contains a different coverage of young growths of various ages in pine, birch and spruce stands. Data describing the share of different species combinations in each forest stand against the same distribution in the total area of 10 forest stands analyzed within the basin area has been used to extrapolate the stand results to give an estimate of the total amount of interception for each catchment:

catchment	1	2	3	4	5
interception [mm] (1985-1999)	125	122	118	124	133

Predictably, the interception in intensively managed woodlands is relatively similar; the greatest difference (between the catchments 3 and 5) reaches 15 mm during six summer months.

The forest stand structure variation within a single group can influence the interception. However this impact is relatively small (Anders and Thomasius 1971). Our measurements show that in pine-birch stands interception depends comparatively little on density. The thinning carried out in these stands has only a minor influence on interception.

If it is necessary, therefore, to evaluate the water-regulatory role of forest over larger areas with forest stands of unknown structure, one regression equation can be used:

$$I = 1.11N + 1,58n - 12 \text{ (mm)} \quad (7)$$

The standard deviation is 10 mm.

Runoff

It is useful to consider separately the hydrological regime of drained forests on deep peat soils and those on hydromorphic mineral soils (Figure 2). The analysis also needs to take into account the significantly different amounts of runoff and groundwater effluent in these areas that are caused by confined aquifer water discharge in areas with a deep peat layer.

The long-term annual runoff from drained peat soils is 425 ± 16 mm, and from drained hydromorphic mineral soils - 245 ± 10 mm respectively, and equates to 53% and 31% of precipitation.

Soil-groundwater system

The hydrological differences between catchments with peat soils and hydromorphic mineral soils are also reflected in the soil-groundwater regime (Figure 3). For example, during the summer months, when the soil moisture is a hindrance for aeration, this decisively influences the functioning of the forest ecosystem. On average, for 10% of the period between May and -October, the groundwater level in areas with drained peat soils is shallower than 35 cm. In areas with drained hydromorphic mineral soils the groundwater level at the same probability is shallower than 65 cm. In dry summers, the probability that the groundwater level falls to 105 cm below ground at drained sites on peat soils is only 10%, but on hydromorphic mineral soils this probability increases to 40%. This means that for nearly half of the growing season there is no ditch water flow ($P_p+P_s \leq 0$).

Water inflow ($P_p + P_s$)

The variations in P_p, as well as P_s at the same site, depend only on the soil groundwater level (H) fluctuation, as the other variables can be considered as constants.

The water inflow ($P_p + P_s$) in catchments with drained peat soils within any one year has fluctuated between –0.01 and 4.92 mm per day (mean=0.9 mm), but in the catchments with drained hydromorphic mineral soils between –0.72 to 3.22 mm per day (mean=0.4 mm). The highest and lowest values of inflow were observed in spring and in summer respectively, both in the catchments with drained peat soils and those with hydromorphic mineral soils. The data obtained show that in the catchments with drained peat soils this component of the water balance totals about 330 mm yearly, accounting for the greater part of the ditch runoff. On average there are 190 mm water flowing out via ditches during the growing season, 8% of which is from precipitation in the growing season, 76% from inflowing ($P_p + P_s$) waters, and 16% from melt-water and spring precipitation.

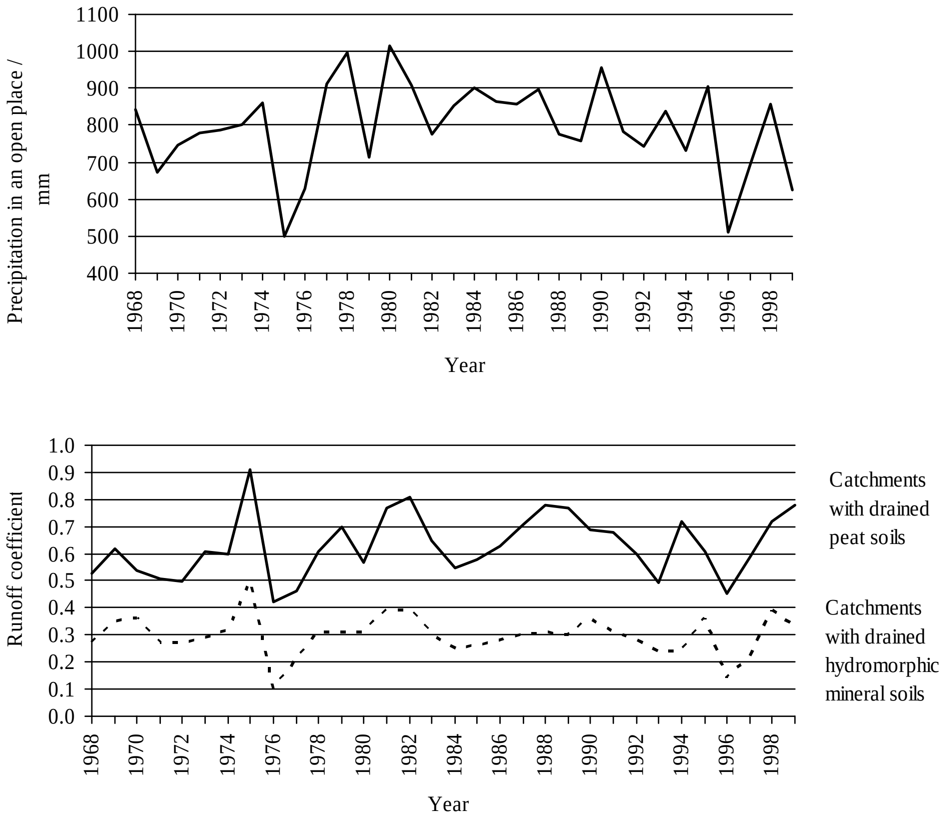

Figure 2. Relation between the runoff and precipitation in catchments of drained peat soils and hydromorphic mineral soils.

Water balance

In conformity with the long-term observations at Vesetnieki, over the period between 1967 and 2003 the water balance equations for the catchments are as follows. For drained peat soils:

$$N\ (797\ mm) + (P_p + P_s)\ (330\ mm) = Q\ (425\ mm) + ET_{V=X}\ (521\ mm) + E_{XI\text{-}IV}\ (181\ mm) \quad (10)$$

For drained hydromorphic mineral soils

$$N\ (797\ mm) + P_s\ (90\ mm) = Q\ (245\ mm) + ET_{V\text{-}X}\ (459\ mm) + E_{XI\text{-}IV}\ (183\ mm) \quad (11)$$

With regard to the consumption component of the water balance, the total evaporation in the growing season (May–October) is calculated as evapotranspiration that is dependent on stand structure parameters according to the method given by Zalitis (1997). The winter season (November–April) is characterized by evaporation from the soil surface, from the tree canopy and from the space under canopy.

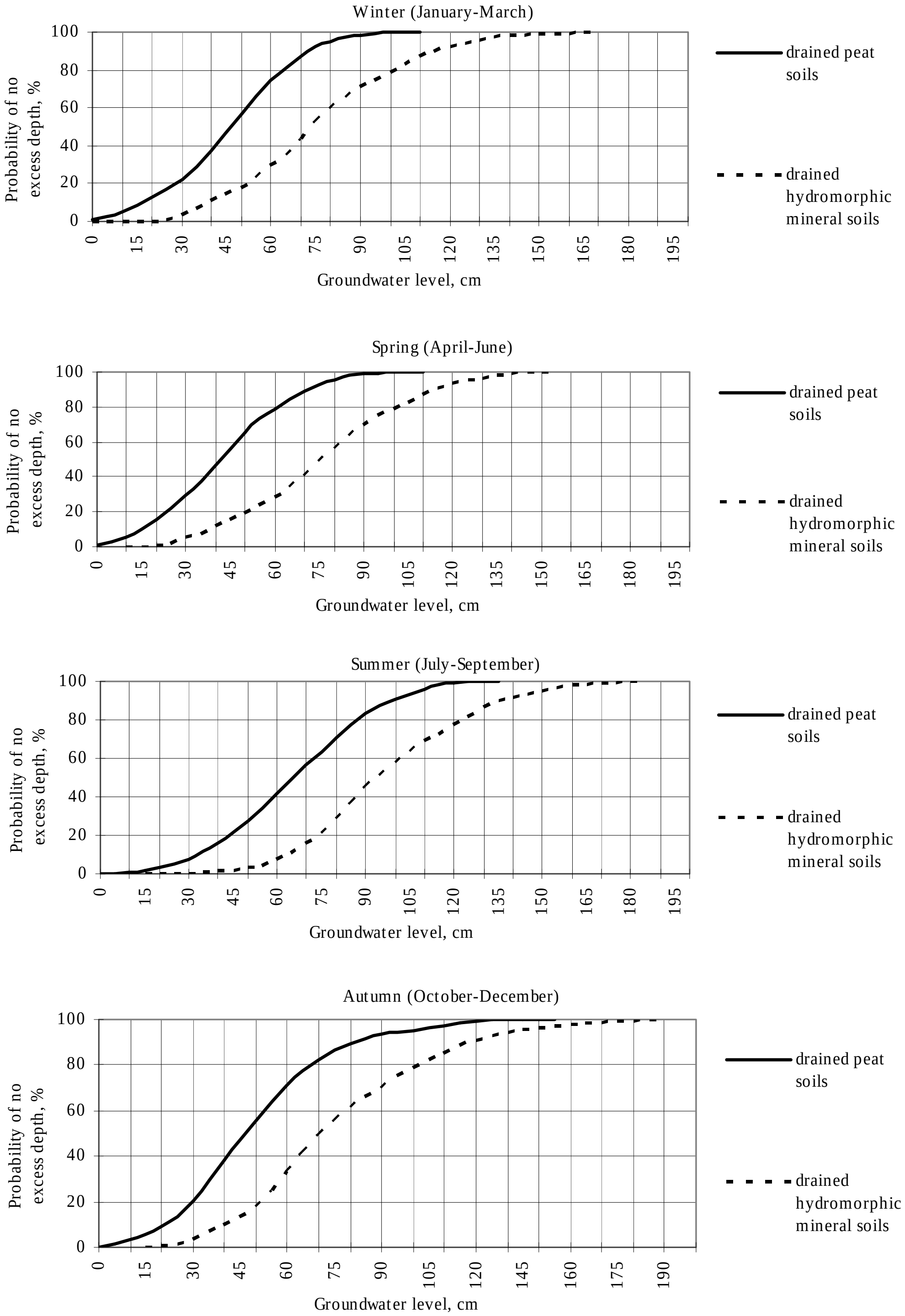

Figure 3. Probability of soil groundwater depth in catchments with drained peat soils and hydromorphic mineral soils in different seasons.

Cycle of biogenous elements

Only in 7 of 120 cases, that correlation between the concentration of biogenous elements in the water and the water runoff amount was statistically significant (Table 1). In all cases, except for N-NH_4^+ in the 2nd catchment in spring, the stated significant relation pointed out to the fact that the concentration of biogenous elements in the water reduces along with increase of the water runoff amount in the ditch. However, the concentrations of biogenous elements mostly have a stochastically fluctuating nature, which allows to consider that together with a larger water amount there is also a larger transport of elements.

By another study in Latvian conditions, it was observed, that the nutrient (phosphorus and nitrogen) concentrations increase during high flow events and decrease during low flow events; it is opposite concerning the water mineralization (calcium and magnesium) (Klavins et al. 2002). According to M. Klavins et al. (2002), relationship between water runoff amount and concentration of potassium is considered as weak. Van Herpe et al. (2000) in the Netherlands have pointed out that, on average, only 32% of the variation in NO_3^- concentration could be explained by the variation in runoff amount in stream, while no significant negative correlations between discharge and concentration could be observed. The significant positive correlation between NO_3^- concentration and water runoff amount was observed for stable or decreasing flow conditions, but to smaller extent for increasing flow conditions (Van Herpe et al. 2000). However, there is no agreement between the present data on the increase in NH_4^+ concentration in the water along with an increase in the water runoff (Ziverts et al. 1996).

Making use of information about the annual water runoff and output of biogenous elements, a model of runoff was made for each examined element in the form of a linear regression equation (Figures 4–9), which is to be used for the determination of the outflow of biogenous elements in the watercourses from the waterlogged forests of Latvia substituting in the equation the value of water runoff from the concerned territory.

The existing data on the outflow of biogenous elements with runoff water in the conditions of Latvia mainly refer to agricultural lands, where it varies between 6.1 and 260.0 kg ha^{-1} $year^{-1}$ for total nitrogen, and between 0.11 and 11.0 kg ha^{-1} $year^{-1}$ for total phosphorus (Skinkis 1992; Jansons 1996; Jansons and Busmanis 2002; Klavins et al. 2002; Apsite and Zvirgzdina 2003). For the calculation of the total nitrogen and total phosphorus export from the different sources, done by Latvian Environment Agency, there were the data from extensive agriculture catchments used for the characterization of background, i.e., forest formed loads – 6.1 and 0.11 kg ha^{-1} $year^{-1}$, respectively (Apsite and Zvirgzdina, 2003).

In our studies, carried out in the drained forests of Vesetnieki Station, for the period from 1969 to 2001, the average runoff of N-NH_4^+ 4.91 kg ha^{-1} $year^{-1}$; N-NO_3^- - 0.51 kg ha^{-1} $year^{-1}$, and P-PO_4^{3-} - 0.17 kg ha^{-1} $year^{-1}$ was stated. By other studies carried out in forested catchments in Latvia the runoff of N-NH_4^+ varied between 0.21 and 1.7 kg ha^{-1} $year^{-1}$; N-NO_3^- between 0.06 and 0.82 kg ha^{-1} $year^{-1}$; P_{tot} and P-PO_4^{3-} between 0.03 and 0.12 kg ha^{-1} $year^{-1}$ (Lyulko and Frolova 1997; Indriksons 2000). The inorganic nitrogen (N-NH_4^+, N-NO_2^-, N-NO_3^-) and phosphorus (P-PO_4^{3-}) are the main forms of the total nitrogen (55% of N_{tot}) and total phosphorus (74% of P_{tot}), respectively (Laznik et al. 1999). Thus, it is possible to conclude, that the runoff loads used by the Latvian Environment Agency only approximately correspond to those measured in the forest lands, and, rather, are slightly underestimated.

The runoff water in the drainage ditches and the soil groundwater in the catchments of Vesetnieki Station is characterized by relatively high average concentration of N-NH_4^+ 1.3 ± 0.09 mg l^{-1} and 3.5 ± 0.27 mg l^{-1}, respectively, which is most likely being characteristic for drained forests in general. According to the surface water quality rates accepted in Latvia ("Virszemes udenu kvalitates prasibas", 1997 [Surface water quality requirements 1997]), the maximum permissible concentration of N-NH_4^+ is 0.39 mg l^{-1}.

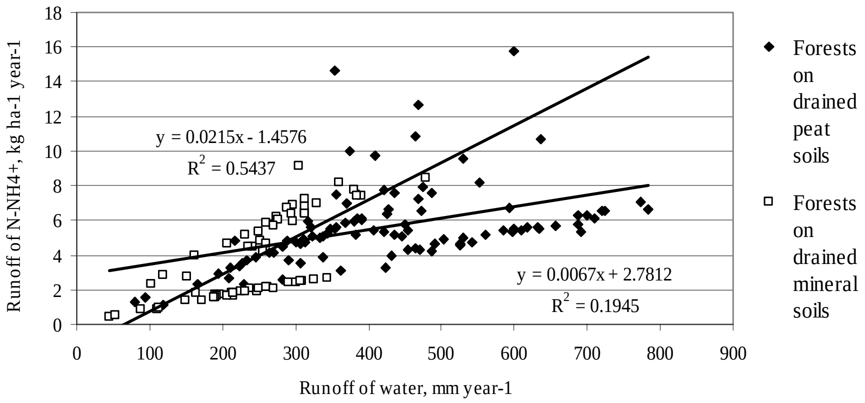

Figure 4. N-NH_4^+ runoff model.

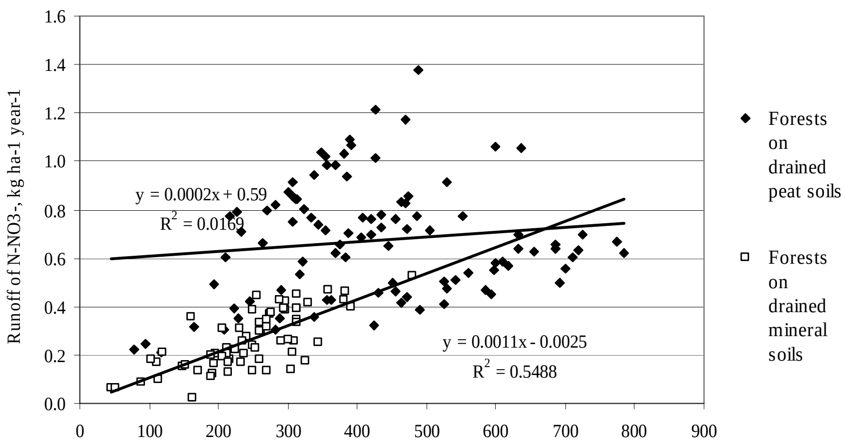

Figure 5. N-NO_3^- runoff model.

The inflow of N-NH_4^+, N-NO_3^-, P-PO_4^{3-} and K^+ obtained at Vesetnieki in the forest ecosystem by the precipitation exceeds the outflow of the same elements by ditch runoff (Table 2). The outflow of Ca^{2+} and Mg^{2+} by ditch runoff is several times higher than the inflow of the respective elements by the precipitation. During the last decade most of forest ecosystems have been recognized as nitrogen sinks accumulating the heightened atmospheric deposits in soil or biomass (Gundersen 1995; Dise and Wright 1995; Johnson and Lindberg 1992). In the different studies nitrogen inflow by precipitation in the forest ecosystem varies between 1.0 and 72.0 kg ha^{-1} $year^{-1}$ and outflow – between 1.3 and 39.0 kg ha^{-1} $year^{-1}$ (Melillo 1981; Nömmik 1983; Matzner 1988; Mohr 1994; Rothe 1997; Dambrine and Ranger 2000). However, there exists some prognosis that, in the future, the nitrogen amount in forests can reach the limit, when an intensive leaching from the soil will takes place (Eichhorn et al. 2001). The loss of nitrogen and other elements except phosphorus from the forested catchment has been obtained by Stevens et al. (1989). The critical load of nitrogen to

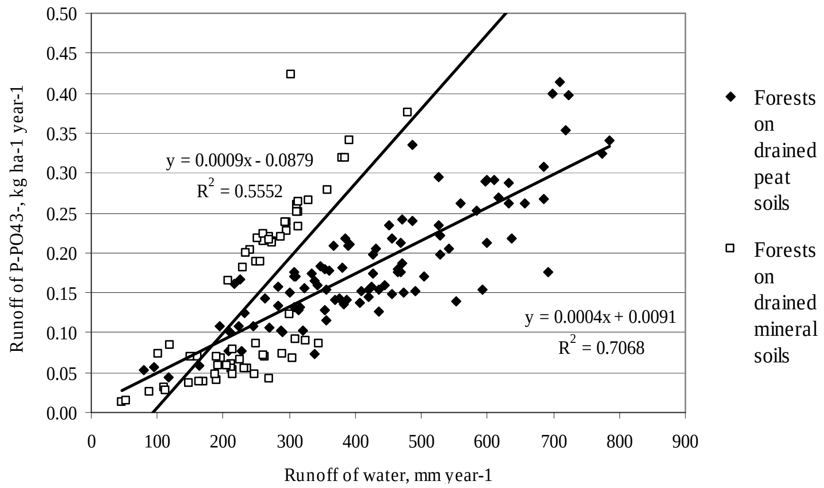

Figure 6. P-PO_4^{3-} runoff model.

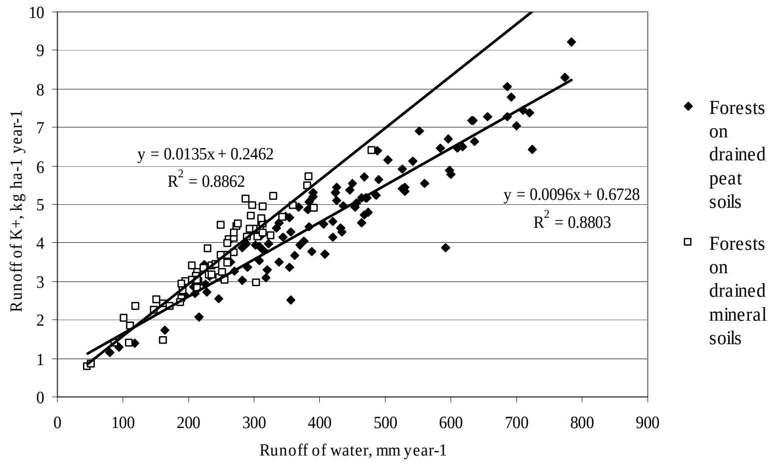

Figure 7. K^+ runoff model.

coniferous forests has been estimated as 10–20 kg ha^{-1} $year^{-1}$ (Bobbink and Roelofs 1995), which already can cause the heightened outflow of nitrates.

The inflow of mineral nitrogen (N-NH_4^+ and N-NO_3^-) by precipitation via throughfall in forest stands at Vesetnieki reaches, on average, 14.3 kg ha^{-1} $year^{-1}$, whereas the average outflow rates in the post-drainage period are 5.45 and 3.71 kg ha^{-1} $year^{-1}$ for the drained peat soils and drained hydromorphic mineral soils, respectively. The present ecosystem nitrogen status at Vesetnieki can be characterized as a nitrogen unsaturated for catchments with drained hydromorphic mineral soils, and saturated to a low extent for the catchments with drained peat soils. This could be explanation for the heightened foliar uptake of nitrogen or its retention in the interception water found at Vesetnieki and by other studies (Lindberg et al.

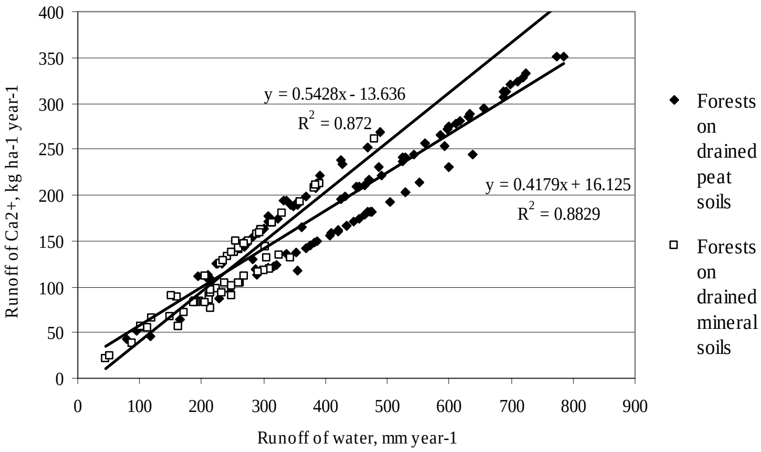

Figure 8. Ca^{2+} runoff model.

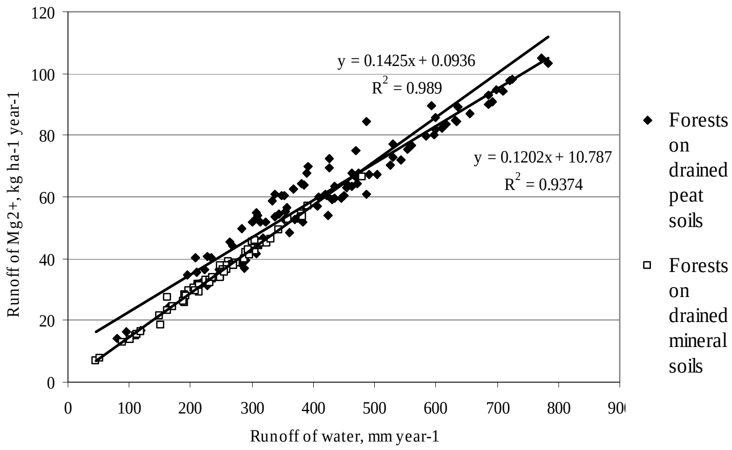

Figure 9. Mg^{2+} runoff model.

1986; Brumme et al. 1992; Marques and Ranger 1997). In case of $N\text{-}NO_3^-$, this retention makes the balance for this ion negative: the outflow via ditch runoff exceeds the inflow by precipitation about 0.17 kg ha^{-1} $year^{-1}$. However, the estimation of complete inflow rates of the elements in forest is very difficult because of variability of the deposit paths (Lindberg et al. 1986; Lovett 1994; Ibrom 1993). The same concerns also the paths of element output.

In Latvian conditions, during the last decade, the total nitrogen deposit varied between 5.43 and 17.42 kg ha^{-1} $year^{-1}$; the wet deposition of $N\text{-}NO_3^-$ varied between 2.03 and 7.98 kg ha^{-1} $year^{-1}$; the wet deposition of $N\text{-}NH_4^+$ - between 2.07 and 8.63 kg ha^{-1} $year^{-1}$ (Lyulko et al. 2002). According to information obtained at the ICP Integrated monitoring plots in Latvia, the average inflow of $N\text{-}NH_4^+$ in an open place by precipitation is 5.4 kg ha^{-1} $year^{-1}$, and 6.0 kg ha^{-1} $year^{-1}$ under the forest canopy (Lyulko and Frolova 1997). These values for $N\text{-}NH_4^+$ are considerably lower, but for $N\text{-}NO_3^-$ – higher as those obtained at Vesetnieki. It could be probably explained by the fact, that methodic of our measurements is different, including also the interception of ground vegetation layer.

Table 1. The average concentrations of the biogenous elements (mg l^{-1}) and the functions to be used for calculations for the five Vesetnieki catchments.

Element	Season	1st catchment	2nd catchment	3rd catchment	4th catchment	5th catchment
$N\text{-}NH_4^+$	Spring	1.69	conc.= 0.0155 + 1.4793 * runoff	0.87	0.94	2.49
	Summer	1.32	0.73	0.64	0.94	2.43
	Autumn	1.48	0.77	1.23	conc.= 1.2284–0.7555 * runoff	2.08
	Winter	1.52	1.02	0.7	0.8	1.48
$N\text{-}NO_3^-$	Spring	0.254	0.236	0.161	0.146	0.199
	Summer	0.073	0.076	0.068	0.100	0.171
	Autumn	0.478	0.208	0.017	0.034	0.041
	Winter	0.163	0.098	0.098	0.036	0.103
$P\text{-}PO_4^{3-}$	Spring	0.079	0.048	0.061	0.029	0.066
	Summer	0.066	0.063	0.116	0.077	conc.= 0.1433–0.1027 * runoff
	Autumn	0.011	0.003	0.008	0.016	0.090
	Winter	0.041	0.036	0.031	0.019	0.091

Table 1. Continued

K^+	Spring	1.51	conc.= 1.3123–0.2808 * runoff	conc.= 1.9184–0.3783 * runoff	1.73	2.20
	Summer	1.48	1.24	1.08	1.58	conc.= 2.3984–1.3092 * runoff
	Autumn	1.13	1.03	0.99	1.10	0.96
	Winter	1.09	1.37	1.23	1.21	1.26
Ca^{2+}	Spring	53.49	39.78	46.71	44.39	55.33
	Summer	72.69	38.52	46.41	46.48	52.08
	Autumn	50.89	36.34	45.49	conc.= 57.696–28.058 * runoff	53.54
	Winter	54.51	38.79	43.33	45.37	55.01
Mg^{2+}	Spring	17.75	15.92	13.44	15.04	13.04
	Summer	20.64	14.56	12.24	16.10	16.07
	Autumn	17.29	14.31	15.31	14.70	15.07
	Winter	14.36	11.61	12.21	12.57	14.12

Table 2. Balance of biogenous elements (kg ha^{-1} year^{-1}).

Element	Time period between 1997 and 2001, average year				Post-drainage period (between 1969 and 2001, average year)		
	Inflow by precipitation		Outflow by ditch runoff			Outflow by ditch runoff	
	In an open place	Via throughfall	Drained peat soils	Drained hydro-morphic mineral soils	Inflow by precipitation in an open place	Drained peat soils	Drained hydro-morphic mineral soils
$N-NH_4^+$	14.8	13.9	4.88	3.50	15.2	5.6	3.8
$N-NO_3^-$	0.90	0.40	0.57	0.21	0.92	0.67	0.26
$P-PO_4^{3-}$	1.39	0.80	0.15	0.13	1.44	0.18	0.14
K^+	11.9	16.2	4.26	2.71	12.2	4.77	3.55
Ca^{2+}	37.8	39.6	170.9	104.6	38.8	193.9	119.3
Mg^{2+}	15.4	16.2	55.0	29.7	15.8	61.9	35.0

Most likely the negative balance of the Ca^{2+} and Mg^{2+} ions is due to the soil groundwater and ditch runoff, supplemented by the confined aquifer water from the upper Devonian dolomite layers (Zalitis and Indriksons 2003). The outflow of Ca^{2+}, Mg^{2+} and N-NO_3^- is significantly higher from forests on a deep peat layer than from those on hydromorphic mineral soils (Table 2).

The yearly total inflow of N-NH_4^+, N-NO_3^- and P-PO_4^{3-} by precipitation is higher in the clearing, while the same for K^+, Ca^{2+} and Mg^{2+} is higher under forest canopy (Table 2). The forest canopies absorb the ammonium more intensively when comparing with nitrate (Potter et al. 1991; Lovett 1992; Piirainen et al. 1998; Stachurski and Zimka 2000), which is characteristic also in our study. Carleton and Kavanagh (1990) observed that the nitrogen foliar uptake increase with the stand age. The throughfall enrichment of potassium, calcium and magnesium has been obtained also by other studies (Ulrich 1983; Bredemeier 1988; Raspe 1997).

The balance of nutrients over a year or longer period is smoothed out, i.e. the inflow does not exceed the outflow. Such an inflow / outflow ratio is a significant precondition for the ecosystem preservation.

4. Conclusions

In the forests on drained peat soils there is significantly higher amount of runoff and a different groundwater regime compared to forests on drained hydromorphic mineral soils.

The increase in interception after drainage, exacerbated by a steadily increasing volume and biomass of standing forest, has an important influence in ensuring favorable soil moisture conditions for the forest growth. The amount of interception from drained forest canopies is approximately 60 mm greater than from undrained forests with low productivity.

The amount of a particular element entering or leaving the ecosystem is controlled more by volume flow than by variability in concentration, so it is possible to use the average values for concentrations of biogenous elements.

The inflow of N-NH_4^+, N-NO_3^-, P-PO_4^{3-} and K^+ to the forest ecosystem with precipitation exceeds the outflow of the same elements in ditch runoff. This suggests that there is an accumulation of these elements in the forest ecosystem. In contrast, the outflow of Ca^{2+} and Mg^{2+} in ditch runoff is several times higher than the inflow of the respective elements with precipitation, which is caused by the confined aquifer water discharge in this area.

The runoff water in the drainage ditches and the soil groundwater in the catchments of Vesetnieki Station is characterized by relatively high average concentration of N-NH_4^+, which do not corresponds the surface water quality requirements and, most likely, is being characteristic for drained forests in general.

There is retention of N-NH_4^+, N-NO_3^- and P-PO_4^{3-} by forest canopy and enrichment of throughfall by K^+, Ca^{2+} and Mg^{2+} obtained, which is connected with the nutrient foliar uptake and leaching, respectively. In case of N-NO_3^-, the retention causes the changes in inflow-outflow ratio.

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Biodiversity and Forest Management

The Effects of Felling Regimes and Silvicultural Treatments on Forest Species with Different Life History Traits: State of the Art and Management Implications

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Abstract

The sustainable management of forest ecosystems implies that silvicultural treatments and cutting regimes do not reduce biodiversity in the long term. The purpose of this synthesis is to analyse the effects of (1) size, intensity and frequency of stand opening, (2) silvicultural treatments upon different elements of forest biodiversity (including dominant trees, understory vegetation, carabid beetles and birds), and (3) to provide practical recommendations for the conservation of forest biodiversity.

Current silvicultural practices tend to reduce the tree diversity at the landscape scale because they truncate the initial and/or final stages of the silvigenetic/successional cycle and are thus detrimental to early- and late-successional trees. The diversity of forest specialists is also partly reduced because large, frequent (in time and/or space) fellings prevent those species from finding a sufficient amount of suitable habitats or having enough time to recolonise after felling. Silvicultural treatments are complementary regarding disturbance regimes, but they are all deficient in terms of key habitats such as small gaps, early- and late-successional phases, senescent trees and deadwood.

Consequently, we recommend: (1) extending rotations and increasing the proportion of early- and late-successional phases, (2) regenerating more often than in lowland French forests today by opening small gaps (<0.15 ha) and (3) continuing regeneration through progressive fellings associated with a green-tree-retention system. Diversifying the silvicultural treatments and increasing the number of biological reserves at the landscape scale are also suggested.

Keywords: biodiversity; felling regime; silvicultural treatment; carabid beetles; birds; vascular and non-vascular plants.

1. Introduction

The choice of the best silvicultural treatment in terms of economic outputs, feasibility of silvicultural practises and monitoring and production of non-market goods is a highly debated issue, which is subject to opposing of the management schools of even-aged and uneven-aged high forest supporters, respectively. In this context, it is often said that forest management practices that produce stands different from natural unmanaged stands in terms of composition, structure and dynamics have a negative impact on forest biodiversity. This statement involves even-aged, monospecific high forest stands and plantations that resemble high-yield agricultural systems (Hansen et al. 1991; Peterken 1996; Kuuluvainen 2002). Other forest management practices such as large clearfellings followed by coniferous plantations are severally criticised by ecologists, scientists and the public partly because they can have a strong impact on soil properties and species composition, but also for aesthetic reasons (Harlow et al. 1997).

In the context of sustainable management of forest ecosystems, we must ensure that our current silvicultural practices (felling methods and silvicultural treatments) will conserve biodiversity in the long term. Felling is a disturbance that modifies the forest at the scale of trees in stands (Spies 1997). The natural succession pattern after felling is well-known for tree species, but is more or less clear for other taxonomic groups (Hunter 1990): "Is the suite of species similar along the different stages of the silvicultural cycle, or are there numerous species which are restricted to specific stages? What are the biological drivers of species diversity throughout the silvicultural cycle?"

The purpose of this review is (1) to analyse short-term and long-term effects of size, intensity and frequency of stand opening, (2) silvicultural treatments on forest biodiversity in terms species composition (including dominant trees, understory vegetation, carabid beetles and birds) and (3) to suggest management practices for the protection of biodiversity. Management recommendations are provided with a confidence level of "very likely", "likely" or "not sure" to protect biodiversity, according to present scientific knowledge and personal opinion. The study groups are: trees, understory vascular and non-vascular plants (except epiphytes), ground beetles (*Coleoptera, Carabidae*) and nesting birds. I will focus on all the species and not especially on threatened species. The role of deadwood for conserving the species is not considered in this paper (see Gosselin (in press) for a review of this issue).

2. Methodology for analysing the effect of forest practices on biodiversity

We selected publications dealing with temperate and boreal forests and did not include tropical literature. The objective is to detect if species can survive under management practices and our analysis was almost exclusively based on species presence and abundance (Bergès in press). An analysis based merely on abundance, richness or diversity of the whole community is insufficient to answer these questions because it does not take into account species' life-history traits (Gosselin and Gosselin in press). It is often useful to separate each taxonomic group into ecological groups based on life-history traits (McIntyre et al. 1995; Lavorel et al. 1997). Trees were separated into early-, mid- and late-successional species groups (Rameau et al. 1989). Understory plants were separated into forest generalists, forest specialists and non-forest species following the habitat preference based on pre-established phytosociological classifications (Rameau et al. 1989; Julve 2002): forest species are defined as species for which forested stands are the most frequent habitat; non-forest species are defined as species for which open lands and ecotones are the most frequent habitat. The distinction between forest specialists and forest generalists are discussed in the text. Carabid beetles are separated into forest specialists, forest generalists and a non-forest species group following the same criteria as those used for plants as proposed by several authors (Niemelä 1999; Koivula 2002; Koivula et al. 2002). Breeding birds were separated into simple groups based on habitat preferences for breeding (Ferry et Frochot 1974; Wiens 1989; Helle and Mönkkönen 1990; Thiollay et al. 1994; Muller 1997; Norton et Hannon 1997), even if more detailed classifications can be proposed that are also based on frequent sites for singing, searching for food or watching for prey or predators (Wiens 1989, Muller 1997, Norton et Hannon, 1997).

Moreover, a complete analysis must include various spatial scales (alpha, beta and gamma diversities) and cover the whole silvicultural cycle (Halpern and Spies 1995; Bergès in press; Gosselin and Gosselin in press).

3. Effect of felling upon biodiversity

The community response of felling depends on size and intensity of the treatment, but also on soil disturbances, site conditions and stand history, i.e. nature and frequency of previous fellings (Pickett and White 1985; Pickett et al. 1987; Lorimer 1989). The intensity of felling is defined as the proportion of harvested trees within the stand and is also linked to soil disturbances (Brunet et al. 1996). Recolonisation success after felling depends on modification of the environment by the dominant species (especially trees), competition between species, species dispersal capacity and seed survival in the seed bank (for plants).

3.1 Trees

Both the size and intensity of a felling influence the composition of tree regeneration and tree growth rate. Shade-intolerant species tend to dominate in large cuts whereas shade-tolerants tend to dominate in small ones (Leak and Filip 1977; McClure and Lee 1993; Taylor et al. 1996).

Tree succession shows the following general trend throughout the silvigenetic cycle following a large disturbance: all species are present at the beginning but early-successional trees (shade-intolerant) dominate first, then decrease and disappear in later stages; mid-successionals dominate during the self-thinning phase and are gradually replaced by late-successional species (shade-tolerant); mid- and late-successional species can coexist in later stages (Rameau et al. 1989).

Silvicultural disturbances create a different disturbance regime compared to natural disturbances: large, intense disturbances are more frequent and stands are often felled before or just after the end of the self-thinning phase depending on rotation length in order to prevent sanitary problems for highly valuable trees. However, it is possible that stands are cut during the understory re-initiation phase (or transition phase) if a late-successional tree species is the objective (i.e. oak-beech forests). Therefore, the late-successional phase (transition and steady-state phases) are very rare at the landscape scale (Niemelä 1999; Spies 1997). Early-successional phases are reduced as much as possible in silvicultural treatments by eliminating early-successional trees. Moreover, the self-thinning phase is avoided by regular, early thinnings, at least in western Europe silviculture (Schütz 1990).

Consequently, fellings have direct effects on tree species diversity: stands dominated by early and late-successional species are dramatically reduced at the landscape scale if a mid-successional species is the objective. This does not necessarily mean that tree species diversity is reduced by management practices at the landscape scale because managers can maintain early- and late-successional species in the understory of stands where mid-successional species are dominant. A reduction species genetic diversity can be expected (Ledig 1992; Hall et al. 1996; Buchert et al. 1997).

3.2 Understory vegetation

Very small gaps (<0.15 ha) or moderate thinnings lead to an increase in abundance and/or species richness of forest species and do not favour non-forest species (Moore and Vankat 1986; Collins and Pickett 1987; Goldblum 1997). Non-forest species colonise more intense and/or larger fellings. Forest species are more or less tolerant to large, intense fellings but this point is still being discussed. Some studies indicate a significant reduction during several years followed by a progressive recovery (Schoonmaker and McKee 1988) while other studies suggest there is only a limited loss of a few forest species, and an increase or no change in the abundance of others (Halpern and Spies 1995; Kirby 1988; Hannerz and Hånell 1997). The forest species group's sensitivity differs among studies and could mainly depend on size and intensity of felling, damage on individuals and soil disturbances such as litter raking, soil compaction, slash deposal or rut formation (Brunet et al. 1996; Deconchat and Balent 2001; Rydgren et al. 1998). The dispersal capacity of these species as well as their non-persistent soil seed bank are also limiting factors for recolonisation (Brunet and von Oheimb 1998; Ehrlén and Eriksson 2000) but colonisation rate assessment is still being discussed (Cain et al. 1998).

The synthesis also shows that the distinction between forest and non-forest plants is rarely satisfactory (Bergès in press). It would be useful (1) to separate forest species into generalists and specialists of late stages by identifying species that are very sensitive to thinning or regeneration felling and/or have a low dispersal capacity and (2) to test if this classification varies with space.

The response of bryophytes to regeneration felling and intense thinning is not always negative, even if their sensitivity to desiccation and direct light make them vulnerable to canopy cover reduction and even more to clear-felling (Økland et al. 1999; Hannerz and Hånell 1997; Beese and Bryant 1999). Indeed, some species are good colonisers of microhabitats such as bare ground or windfall stem end disks (Hannerz and Hånell 1997; Jonsson and Esseen 1998; Haeussler et al. 2002). Other studies show that bryophyte richness and diversity are higher on disturbed areas. Further studies are needed on bryophyte succession, but it seems to differ from the vascular plant succession as abundance peaks during the self-thinning phase when vascular plant abundance is minimal and remains high during late-successional stages (Alaback 1982).

Consequently, I propose four predictions about the short-term response of groups of plants to felling intensity and size (cf. Figure 1) and four corresponding predictions about long-term changes throughout the silvigenetic cycle (cf. Figure 2). Species can be separated into four ecological groups: non-forest species, forest generalists, forest specialists and bryophytes. Forest generalists are species that are more frequent in closed-canopy forests but that are not restricted to a specific successional stage; forest specialists are forest species that are more frequent in or are exclusive to late-successional stages. One prediction is proposed for each group. Moreover, it is necessary to define these groups before testing these assumptions.

Short-term group responses to felling size and/or intensity are:

1. non-forest species are absent in small gap and standard thinnings then rise sharply and plateau after a second threshold disturbance intensity/size;
2. forest generalists increase as disturbance intensity/size increase and then plateau;
3. forest specialists are not affected by felling until the second threshold value is reached (progressive cut) and then they decrease;
4. most bryophytes respond negatively to felling.

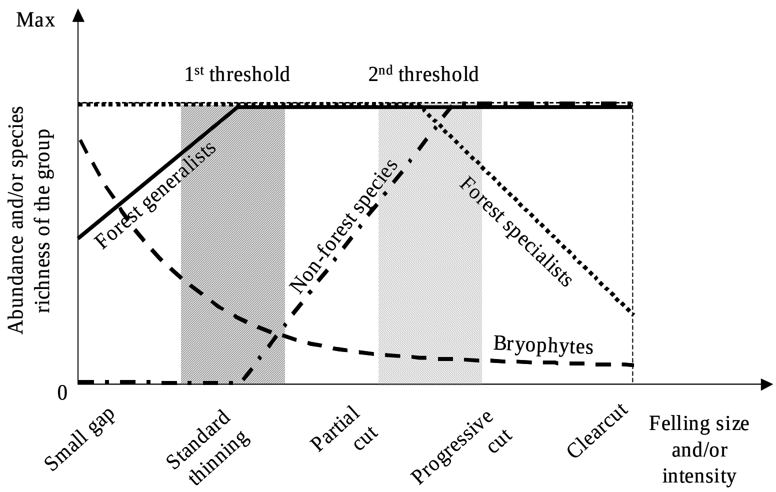

Figure 1. Predicted changes in abundance and/or species richness of the four ecological groups of understory vegetation according to felling size and/or intensity.

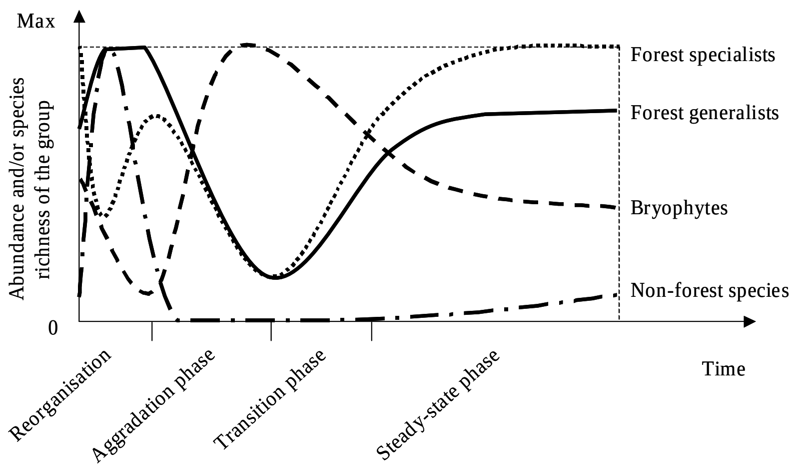

Figure 2. Change in abundance and/or species richness of the 4 ecological groups of understory vegetation throughout a natural succession following a large, intense disturbance. These changes are general trends and most of them have yet to be tested. The 4 different phases of the silvigenetic cycle are described by Bormann and Likens (1979), Oliver and Larson (1996) or Spies (1997).

Long-term changes predictions throughout a silvigenetic cycle following a large, intensive disturbance are:

1. non-forest species appear at the beginning of the cycle and then decrease after canopy closure;
2. forest generalists are slightly favoured by felling and increase in abundance and/or richness, then decrease gradually during the self-thinning phase until the canopy opens in the transition phase; they increase again at the end of the cycle;
3. forest specialists decrease after felling then recolonise but decrease again gradually during the self-thinning phase and display more or less the same pattern as forest generalists at the end of the cycle;
4. many bryophytes display a different pattern: they decrease after felling then increase during self-thinning and then decrease again at the end of the cycle. Other bryophytes can be favoured by felling but are not indicated on Figure 1 and Figure 2.

3.3 Carabid beetles

Carabid beetles' response to felling is close to the response of understory vegetation. I will only describe general trends most of which have already been widely discussed (Niemelä et al. 1993; Haila et al. 1994; Spence et al. 1996; Atlegrim et al. 1997; Butterfield 1997; Niemelä 1997; Fahy et Gormally 1998; Abildsnes and Tømmerås 2000; Heliölä et al. 2001; Koivula 2002). Thinnings and small-gap cuts are not likely to have a positive impact on forest species; however, this point has been little documented (Atlegrim et al. 1997; du Bus de Warnaffe 2002; Koivula 2002). Response to clear-felling is largely documented in boreal coniferous forests but less so in temperate ecosystems (Niemelä et al. 1993; Pajunen et al. 1995):

1. the stands are quickly colonised by non-forest species (Jennings et al. 1986; Baguette and Gerard 1993; Niemelä et al. 1993; Koivula et al. 2002);
2. forest generalist species remains stable, quickly recolonise or even increase in abundance and/or richness after clear felling (Baguette and Gerard 1993; Niemelä et al. 1993; Koivula et al. 2002);
3. forest specialists that require specific site conditions or prey related to closed stands decrease more or less rapidly and strongly after felling (Lenski 1982; Heliövaara and Väisänen 1984; Niemelä et al. 1993; Koivula et al. 2002).

Community composition shows dramatic shifts between young and old stages of the silvicultural cycle (Niemelä et al. 1993; Baguette and Gerard 1993; Spence 1997):

1. open habitat species that increase substantially after felling in abundance and/or richness (see above) decrease after canopy closure (Szyszko 1991; Baguette and Gerard 1993; Haila et al. 1994; Butterfield 1997; Ings and Hartley 1999; Koivula et al. 2002);
2. forest generalists can be abundant in early stages and then decrease as the stand closes but their richness is quite stable throughout the cycle (Koivula et al. 2002);
3. several forest specialists are absent or rare in young stages (5 to 10 years after clear-felling) but recover during the end of the cycle (Baguette and Gerard 1993; Jukes et al. 2001; Koivula et al. 2002).

3.4 Nesting birds

Nesting birds abundance/richness is generally positively affected or unaffected by thinning or small gap creation. However, as the relationship between stand basal area and abundance or species richness of birds is curvilinear (Blondel et al. 1973; James and Wamer 1982), the effect on bird diversity may depend on stand basal area before felling but also on the vertical structure, tree height, species, amount of old trees, and amount of dead wood.

Species composition is modified after an intense felling but, due to their great mobility, bird local extinction, colonisation and recolonisation are usually very rapid processes, depending mostly on habitat suitability. However, if fellings is practised over very large areas, species can be extirpated. According to the publications reviewed and based on the ecological groups of species proposed by the authors, four predictions can be proposed to describe short-term responses of avifauna to two types of fellings (partial or small-gap felling versus clearfelling) but contrary to understory vegetation and carabids, these assumptions concern community and ecological groups responses:

1. open-habitat species increase in abundance and/or richness after felling and forest species decrease as felling intensity increases (Norton and Hannon 1997; Chambers et al. 1999; Beese and Bryant 1999; Tittler et al. 2001). Therefore, species richness and/or abundance decrease(s) from untouched stands to partial felling and clearfelling; however, this result also depend on the spatial scale, especially the patch size.
2. the abundance and/or richness of old-forest bird species are retained fairly well in partial felling compared to untouched stands but decrease substantially in clearfelling whereas open habitat species maintain similar population levels in partial and clearfelling (Ferry and Frochot 1970; Ferry and Frochot 1974; Muller 1995; King and DeGraaf 2000). Therefore, the abundance and/or species richness is higher in partial felling compared to control and clearfelling. Species composition in regeneration felling is intermediate between composition in control and clearfelling (Baguette et al. 1994; DeGraaf et al. 1991; Muller 1995; Annand and Thompson 1997; King and DeGraaf 2000);
3. the same pattern as prediction (2) but with colonisation in partial fellings of birds typical of open stands with a few residual trees (Ferry and Frochot 1974; Hansen et al. 1995). Abundance and/or richness is higher in partial felling compared to untouched or clearfelled stands. Maximum abundance for these species corresponds to 4–15 trees per ha according to Hansen et al. (1995), but also to more closed stands according to King and DeGraaf (2000);
4. forest species group do not vary in abundance/richness but open-habitat species increase as felling size/and or intensity increases (Costello et al. 2000). So, bird abundance and/or species richness increase with felling intensity, which disagrees with prediction (1).

The composition and diversity of birds also display dramatic shifts during the silvicultural cycle (Bosakowski 1997) and the following general pattern can be proposed (Helle and Mönkkönen 1990; Muller 1995):

1. abundance and/or species richness of open-habitat birds that nest on the ground or shrubs reaches a maximum during early stages (sapling/thicket), then this group disappears but comes back to some extent in late stages when canopy gaps are created (old high forest);
2. species that nest on shrubs and trees increase during the cycle and reach their maximum in mature stands; but some authors also mention some “forest generalists” that are present at all stages but peak in young or intermediate stages (thicket, pole or young high forest) (Muller 1995); other authors distinguish species that prefer stratified or unstratified mature stands;
3. forest hole-nesting species are present only in late stages due to the lack of senescent trees in earlier stages for most species (old high forests).

3.5 Conclusions

3.5.1 Positive effect of thinning (except for some bryophytes)

Thinnings are necessary for the maintenance of vascular plants, at least for non-boreal species: when practised early in the rotation they can create earlier favourable conditions for herb layer and shrub recovery, provided that rotations are long enough for a significant development of shrub layer before the final felling (Kirby 1988). Low-intensity thinning regimes and conservation of dense intermediate and upper layers in mature stands for 50 to 100 years may not provide adapted conditions for understory vegetation (except for many bryophytes) and many animal species and are likely to threaten their survival in the long-term (Chevalier 2003).

3.5.2 Likely positive effects of small gaps and large partial fellings

The felling size and intensity above which forest species and non-forest species respectively decrease and increase are poorly known and depend on the taxonomic groups under consideration, site conditions and site history. First, it is accepted that forest species generally do not suffer from and sometimes are favoured by very small gaps (0.15 ha) whereas non-forest species do not colonise these gaps. However, an advantage of a large partial felling compared to a large clearfelling has not been demonstrated for understory vegetation and carabids, i.e. it is not sure that typical forest species are maintained in the long term. Partial fellings that keep a few single or grouped trees until their death are interesting for birds, especially because these stands can provide suitable habitats for some specialised forest species (Ferry and Frochot 1974, Hansen et al. 1995).

3.5.3 Successional pattern during silvicultural cycles for the different taxonomic groups

Light, water and nutrient availability (largely controlled by tree composition and density) and plant and carabid species dispersal capacity are the main drivers of understory vegetation and carabid succession (Halpern 1989). Stand composition and structure are the main drivers of bird succession. Moreover, understory vegetation succession following clearfelling approaches an "initial floristic model" succession (Egler 1954) because the different species groups are present together at the beginning of the succession. Bird communities follow more a "relay floristic model" succession (Egler 1954) since there are many early- and late-successional species, even if several species are generalists. Carabid succession is close to a "relay floristic model" succession but the late-stage species that are absent from the early stages might be limited by a high sensitivity to disturbance combined with a low dispersal capacity and not by habitat suitability per se.

Even if succession patterns following a large disturbance differ among taxonomic groups, they do share similar trends (du Bus de Warnaffe 2002): (1) a high quantity of mobile to very mobile species in the young stages (annual and anemochorous plants, macropterous carabids and migratory birds), (2) a high quantity of not very mobile species (geophytes, brachypterous carabids and sedentary birds) in the mature stages; this is less clear for understory vegetation and carabids and (3) reduced total abundance and species richness in the intermediate stages.

3.5.4 Long-term extinction risks

The short-term and long-term responses of species depend on the felling regime. For trees, early and late stages have to be preserved because they provide abundant seedling sources for early- and late-successional species. For carabids and understory vegetation, forest specialists are negatively affected by intense, large and frequent in time and space fellings. Regeneration size and process as well as rotation length have to be adjusted to their life-history traits (ecological requirements, sensitivity to soil disturbance, dispersal capacity) in order to prevent these species from reducing too much in abundance after felling and to give them enough time to colonise the stand once suitable habitat is recreated and before the next regeneration felling. For birds, felling regime is not likely to strongly threaten forest avifauna biodiversity provided distinctive elements of late stages are present at the landscape scale and landscape fragmentation does not increase (Muller 1997; Muller 1999). The first priority is to conserve large and/or senescent trees because hole-nesting birds can only breed there (Sharpe et al. 1996). However, it is not easy to specify the appropriate amount of these trees at the landscape scale.

3.5.5 Role of open space habitat for open-habitat forest species

Within-forest open habitats also play a role in conserving species of early and mid-successional trees that can only regenerate through medium-to-large gaps (e.g. according to Bary-Lenger and Nebout (1993) a gap size over 0.3 ha is necessary to regenerate *Quercus petraea* Liebl.). These habitats are also important for edge-forest understory plants and non-forest plants (Peterken and Francis 1999) but also for some carabids (Desender 1986) and birds (du Bus de Warnaffe 2002).

4. Role of silvicultural treatments

4.1 Theoretical approach

Silvicultural treatments mutually complement each other in terms of general disturbance regime and several ecological gradients (composition, horizontal and vertical structure) but they are all undersupplied in terms of other key-habitats for species adapted to naturally dynamic forests such as very small gaps, long cycle duration, amount of pioneer and late-successional phases, large and senescent trees and deadwood volume (Gosselin in press).

we can discuss what kind of disturbance regime management should imitate to conserve biodiversity. For some authors, species have adapted to the natural disturbance regime (Hansen et al. 1991; Perry 1994; Angelstam 1996; Schnitzler and Borlea 1998): “It is likely that a human-related disturbance regime has reduced the abundance of species that are typical of habitats that have become rare or absent at the landscape scale; these species will be positively favoured if management practices increase the amount of suitable habitats. Indeed, some key elements related to biodiversity are missing in our different silvicultural treatments”.

A second hypothesis is: “species could have adapted to disturbance regimes related to past silvicultural treatments and that any dramatic change in treatment regime would lead to a generalised mid- or long-term decrease. From an evolutionary point of view, it is possible that the species least tolerant to human management are now extinct and the most tolerant species now dominate in our forest ecosystems" (Rackham 1980; Peterken 1981; Barkham 1992).

A third hypothesis is: "It may therefore be reasonable to provide species with a variety of disturbance regimes (thanks to various silvicultural treatments) because the disturbance regime to which the species are adapted is unknown, especially in the context of the unpredictable impact of global change upon biodiversity"" (Bergès et al. 2002).

4.2 Bibliographic synthesis

The theoretical approach can be completed by an analysis of published results on French forests. Unfortunately, the comparison of different silvicultural treatments is generally partial and often debatable (see however Thiollay et al. (1994) or Chevalier (2003) for rigorous comparisons).

4.2.1 Trees

Coppice treatment has favoured tree species capable of vegetative regeneration whereas high forest (even and uneven-aged) encourages highly valuable species using sexual regeneration. Moreover, short-rotation treatments (coppice and coppice-with-standards) have always favoured early-successional tree species whereas the other silvicultural treatments currently in use benefit mid-successional trees and to a lesser extent late-successional trees (Becker 1979).

4.2.2 Other species groups

The results are less documented for the other groups, except for birds (Muller 1999, Thiollay et al. 1994). The conclusions are: (1) a few species are only present in one of the treatments (coppice or coppice-with-standards); (2) most species are present in both treatments but distribution over space and time varies (Muller 1999).

A negative long-term effect of even-aged high forest treatment on some vernal geophytes and shade-tolerant ant-dispersed perennials, bryophytes, forest specialist carabids and hole-nesting birds might be expected. This negative effect of even-aged high forest could be reduced by regenerating through progressive fellings in association with a green/dead-tree retention system. A negative effect of very closed canopy and high stand volume (which is the case in even-aged high forest treatment) on many species and especially on vascular plants can also be predicted (Chevalier 2003), but may be moderated by opening small gaps (>0.05 ha) in mature stands at least 50 years before regeneration.

5. Practical management recommendations

5.1 Felling regime

The priority is to provide habitat continuity for forest specialists or at least to give them enough time for stand recolonisation. This involves changing management practices at the landscape scale in 5 ways:

a) Extending the silvicultural cycle to 150–300 years if possible on at least 25% of the total surface area would very likely favour forest specialists (Curtis 1997, Ferris et al. 2000, Gosselin in press).
b) Using natural regeneration by small gaps (<0.15 ha) or by progressive felling over large areas and retaining large trees or small groups of trees until their death (green-tree retention system) is likely to favour forest specialists (Halpern and Spies 1995; Curtis 1997; Ferris et al. 2000).
c) Preserving old stands close to recently regenerated ones through an adapted spatial organisation of the fellings is likely to favour forest specialists (Niemelä 1999; Koivula et al. 2002). This recommendation is difficult to implement in the field and specific studies should be devoted to this issue.
d) Keeping a large amount of undisturbed understory layer after fellings or thinnings is likely to favour forest species recolonisation from inside (Brunet et al. 1996). The target proportion would be at least 50% of undisturbed understory layer but this point still needs to be specified by further studies dealing with the impact of mechanical harvesting and soil disturbances on forest specialists.
e) Continuing medium-intensity thinnings but with varying periodicity is very likely to favour a majority of forest species, except many bryophytes.

Another recommendation concerns open-habitat species:

f) Keeping a limited number of well-distributed large clearcuts (>2 ha) is likely to maintain several forest species (du Bus de Warnaffe 2002) but even more open-habitat species (early-successional trees, edge-habitat and non-forest plants, open-habitat carabids and birds).

5.2 Silvicultural treatments

Structuring elements of biodiversity reveal considerable differences among different silvicultural treatments (disturbance regime, cycle duration, tree species composition, tree biomass, senescent trees and deadwood density). Consequently, the following recommendations can be made:

g) Increasing the proportion of large stands (>20 ha) that correspond to early and late stages of silvigenetic cycles is very likely to favour many species, especially the early- and late-successional trees but also all the species restricted to these phases (Gosselin in press). This implies increasing the number and the total area of biological reserves and having more pioneer trees-based regeneration instead of immediate regeneration with a commercial tree species.
h) Varying the silvicultural treatments at the landscape scale to ensure a variety of disturbance regimes and tree successions is very likely to favour biodiversity (Hunter 1990; Hansen et al. 1991; Perry 1994; Angelstam 1996, Schnitzler and Borlea 1998; Spies and Turner 1999; Kuuluvainen 2002). This implies favouring small gap-regeneration silviculture (e.g. group-selection high forest) but without standardising gap size.
i) Preserving coppice and coppice-with-standards treatments may not be necessary for biodiversity purposes even if coppice-with-standards preserves large trees throughout the entire cycle. The latter are mainly transitory stands resulting from abandoned treatments. Species could have adapted to disturbance regimes related to past silvicultural treatments and that any dramatic change in treatment regime would lead to a generalised mid- or

long-term decrease. From an evolutionary point of view, it is possible that the species least tolerant to human management are now extinct and the most tolerant species now dominate in our forest ecosystems (Rackham 1980; Peterken 1981; Barkham 1992).

6. Conclusions

Maintaining biodiversity would be a complex problem even if our knowledge of the impact of forest management on plant and animal diversity was better. Species have diverse and sometimes opposite ecological requirements. To answer the question: "Does a given practice have a positive or negative impact on biodiversity?", it is necessary: (1) to separate species into ecological groups (2) to define judgement criteria by stating the importance of certain ecological groups (like forest specialists are considered more important than open-land species) (3) to test if the various responses of the individual species are reliably summarised by the responses of theses groups and (4) to analyse local and gamma diversities (Gosselin and Gosselin in press; Thiollay et al. 1994; Halpern and Spies 1995) and how they are affected by landscape-scale factors. Several studies show that different fellings and/or silvicultural treatments are complementary (du Bus de Warnaffe 2002). The impact of clearfelling cannot be considered to be totally and systematically negative, and even it is needed for some species suffering from too little disturbance, but long-term risks for early- and late-successional trees and plant and animal old-forest specialists are specified. However, no treatment or type of felling should be completely excluded: but a landscape-scale balance that is representative for the disturbance regime is necessary.

The recommendations that I propose confirm the interest of some classical management practices (e.g. thinnings) for conserving many species but also stress the short-comings of current management practices, which do not encourage key structural elements of naturally dynamic unmanaged forests. The main suggestion is a precautionary principle: "To diversify disturbance regimes within a territory and maintain these structuring elements". This should ensure ecosystem variety and thus maintain biodiversity (Hunter 1990). Contrary to the European and North-American boreal forests, I do not think that reproducing a natural disturbance regime through management is the only solution because the management history of our forest ecosystems also has to be taken into account. It would be reasonable to provide species with a variety of disturbance regimes (thanks to various silvicultural treatments) because the disturbance regime to which the species are adapted is unknown, especially in the context of the unpredictable impact of global change upon biodiversity (Bergès et al. 2002).

Lastly, I must insist on several crucial needs (Bergès et al. 2002): (1) additional research to gain new information on species life-history traits, especially their capacity to resist or to adapt to disturbance (Halpern and Spies 1995; Roberts and Gilliam 1995); (2) long-term monitoring programs in biodiversity to assess changes in biodiversity at a landscape scale and (3) development of new management approaches (e.g. through adaptive management).

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Biodiversity in the UK's Forests – Recent Policy Developments and Future Research Challenges

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Abstract

Afforestation in the United Kingdom in the latter half of the twentieth century was seen by many to be the antithesis of conservation. Many of the new forests were created to produce timber by planting exotic tree species and there was criticism of the potential loss of open ground habitats and associated species. Policy and practice has now changed substantially and biodiversity is an important objective – most recently articulated in the individual Country forestry strategies. Protection of important woodland and open-ground habitats is more secure, and habitats are being restored where the afforestation was inappropriate. Recent research has provided evidence that the new forests are becoming habitats with significant biodiversity value. A number of tools and pieces of advice spanning a range of spatial scales have been produced to guide forest and land managers. The challenge for UK sustainable forest management is to build on these developments. The knowledge needed to deliver the new Country policies is analysed. Many of the current policy aims have an inadequate evidence base, and this also hinders the specification of delivery mechanisms. A number of specific research needs are identified. There is a general requirement to develop more integrated models and to better understand the transference of appropriate knowledge from other studies and other countries.

Keywords: forest policy; planted forests; landscape ecology; research needs.

1. Introduction

The UK lost much woodland, and by association much of its woodland biodiversity, prior to the start of the twentieth century. Post-glacial colonisation probably resulted in approximately 75% of the land cover being wooded with an assemblage of tree species restricted by the

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timing of the loss of the land bridge to Europe. The range of native conifers is particularly limited compared to continental Europe (with no *Picea*, *Abies* or *Larix* spp). Man rapidly began to clear forests and it is estimated that woodland cover may have been as low as 15% in England in medieval times, and was probably around 4% in Scotland by the 17th Century. Many notable members of the woodland fauna such as wolf, bear, and beaver were lost in the period 900–1700 A.D. Afforestation commenced in the 18th Century, but there was further extensive felling during the First World War. Substantial afforestation then commenced and as a result there were dramatic changes in the area and composition of forests in the UK during the last century. The area of woodland in the UK increased from around 5% cover at the start of the 20th century to over 11% at the start of the 21st century (Table 1). Many of the new forests have been planted using exotic species of North West American origin. For example in the most recent National inventory, approximately 49% of the forest area was coniferous, and of this 49% was of Sitka spruce (*Picea sitchensis*).

The policy drivers behind afforestation and associated objectives of forest management have changed during the period – due to domestic and international changes in policy, timber supply and profitability. The initial thrust of the afforestation was timber production, and exotic species were chosen because of their superior yield – and the depauperate nature of the coniferous tree flora of the UK. A combination of domestic pressures in the 1980's (budget changes, loss of valued open ground habitat, legal protection), and then the global and Pan-European moves of the 1990s have revolutionised the goals and to some extent the practice of UK forestry. A focus of conservation concern on special non-forest (and some native woodland) sites has been supplemented by a general attention to biodiversity – and most recently the ambition of multi-purpose, sustainable forestry. Of course, the visibility of these changes in the character of the forests does lag. The interaction of the elements of sustainability is particularly pertinent in a highly populated country with a long tradition of land (though little forestry) management. The lack of natural woodland, and unique maritime position, create particular difficulties in assembling a body of evidence on which to base policy and practice – and suggest that there may be difficulties in slavish adherence to standards developed elsewhere. The purpose of this paper is to:

1. Briefly review the way in which nature conservation and biodiversity have become embedded in UK forest policy and practice;
2. Describe recent research findings and new policy developments, analysing the extent to which one is supported by the other
3. Identify gaps in the evidence or knowledge base and propose new research priorities

Table 1. Change in Woodland area in UK. Adapted from information in UK Sustainable Forestry Indicators (Forestry Commission 2002b).

	Woodland area (thousand hectares) %						% woodland cover
Year	1924	1947	1965	1980	1995–99	2002	2002
England	660	755	886	948	1097	1104	8.5%
Scotland	435	513	656	920	1281	1324	16.9
Wales	103	128	201	241	287	288	13.9
N Ireland	13	23	42	66	81	84	6.2
UK	1211	1419	1785	2175	2746	2800	11.5%

Sources: GB Censuses of Woodland 1924 to 1980, NIWT 1995–1999, and NI Forest Service

In doing so, we also aim to provide an update on the approach to biodiversity conservation previously provided in an EFI publication by (Hodge et al. 1998).

2. Review of recent developments in policy, practice and research

In the UK it is generally agreed that biodiversity can benefit from attention to 3 types of woodland - ancient and semi-natural woodlands (e.g. focus on restoration), creation of new native woodlands, and enhancement of commercial forests. That forest biodiversity is worthy of political concern and public support is underlined by a recent UK forest visitor survey (Forestry Commission 2003) indicating that viewing wildlife is a strong incentive for recreation in the woodlands.

There have been substantial changes in policy and practice relating to biodiversity and conservation. The main policy milestones are summarised in Table 2.

It is clear that policy has developed in response to a variety of domestic and international drivers of change. Five mechanisms have been identified in the subsequent delivery of policy into practice (Rollinson 2003):

1. Research & Inventory – to provide an informed basis for delivery
2. Standards – to set the requirements for good forestry practice
3. Guidance – to encourage adoption of best practice
4. Regulations – to protect the environment and control potentially damaging operations
5. Incentives – to encourage adoption of new programmes

Progress will be summarised against each of these, and then their impact on practice will be briefly reviewed.

Research and Inventory

Researchers have responded to the changing agenda – moving from research programmes looking at the injurious aspects of wildlife in forests, to nature conservation values of native habitats and key species groups, and most recently the biodiversity value and potential of the new forests. The current research programme incorporates biodiversity assessment, forest habitat management, landscape ecology the needs of special species, and genetic conservation.

The main recent focus has been the basic assessment of biodiversity within planted forests (Humphrey et al. in press). This has resulted in a raised awareness of the value of plantations of introduced conifer species as habitats for native flora and fauna (Humphrey et al. 2000; Humphrey and Quine 2001; Humphrey et al. 2002a), an understanding of the factors influencing assemblages, and the need for improved integration with native woodland and other habitats at the landscape scale (Humphrey et al. 2003). Reviews have also considered the relevance to UK forests of concepts developed elsewhere e.g. mimicking of natural disturbance regimes (Quine et al. 1999).

Enhanced monitoring of the basic woodland resource (National Inventory of Woodlands and Trees) incorporating structural features of biodiversity significance is now in place, as well as monitoring of specific indicators (e.g. headline indicators of sustainability including woodland birds) (Forestry Commission 2002b).

Standards and Guidance

The UK Forestry Standard sets out the government policy on sustainable forest management, and provides a statement of the standards deemed to be good practice (Anonymous 1998).

Table 2. A brief history of forest policy relevant to biodiversity development conservation and enhancement, with an emphasis on the last 20 years.

Milestones in development of the approach to biodiversity conservation		Supporting comments
1919	Forestry Act – the first forest policy establishes the Forestry Commission	Emphasis on creating a strategic reserve of timber
1945	Forestry Act introduces aid for and promotion of private forestry as well as state forests.	Restates target of increasing the productive forest area
1957	Policy review gives prominence to economic returns from forestry	Leads to replacement of broadleaved woods with conifers
1985	Wildlife and Countryside Act (Amdt) requires Forestry Commission to seek a reasonable balance between afforestation/timber production and nature and landscape conservation	Broadleaves policy was initiated
1988	Removal of tax relief on afforestation and ban on coniferous afforestation in English uplands	Reduced fiscal support for exotic plantations
1991	Restatement of Forestry Policy for GB aims for 'The Sustainable management of our existing woods and forests'	Explicit broadening of policy objectives
1994	'Sustainable Forestry: the UK Programme' and 'Biodiversity – the UK Action Plan' published	Response to Rio (1992) and Helsinki (1993) agreements
1998	UK Forestry Standard published	Provides standard for practice of sustainable forestry
1998	Forestry Commission policy statement on 'Biodiversity and forestry'	Establishing key elements of internal FC policy
1998	– England Forestry Strategy	First of the country policy statements
1999	UKWAS criteria published	Voluntary standard for certification
2000	Scottish Forestry Strategy	Country statement responding to devolved politics
2000	– Countryside Rights Of Way Act – England and Wales	Additional protection to special sites
2001	Welsh forestry strategy	Country statement responding to devolved politics
2001	N Ireland Forestry Strategy	Country statement responding to devolved politics
2002	CBD meeting – Johannesburg	
2002	UK Sustainable Forestry Indicators	To monitor progress against the Standard
2003	Sustainable forestry in the UK: the UK's National Forest Programme	Draws together the 4 country forestry strategies and the UK Forestry Standard
2003	England Biodiversity Strategy	General govenrment strategy on biodiversity

Supporting guidance is given as a result of applied research and recent developments in specifying practice are summarised in Table 3. They have included increased use of an ecological site classification (to ensure a better fit of tree species or restored woodland community to the potential of the site); better understanding of the needs of some rare species (both woodland e.g. red squirrel, and non-woodland e.g. golden eagle); techniques for the restoration of special habitats; and the first steps towards a more integrated landscape approach. Some of this has been supported entirely by UK-based research, some by careful transference of findings from overseas – and some by breath-taking leaps of imagination! Literature from the Pacific North West of America (and to some extent the boreal forest of Scandinavia) has been particularly influential – not just in steering researchers, but in informing pressure groups (environmental non-government organisations) and others engaged in debate over plans, policies, standards and certification.

Regulations, Incentives (and intervention)

There is comprehensive protection for special sites (e.g. SSSI's, SAC's, SPA's) meeting the needs of European and domestic legislation using a variety of instruments including felling licences and statutory consultation. UK came out highly in recent WWF league table on improvements on forest protection (WWF European Forest Protection Programme 2003), but there are still concerns over losses of special habitats.

Grant aid is given to private owners to encourage sustainable forest practices – including the conservation and enhancement of biodiversity. A recent review of the economics of government intervention in forestry in England has indicated that measures to enhance and

Table 3. Examples of recent guidance relating to biodiversity management (adapted from Humphrey et al. In press)

General theme	Topics	Publications
Needs of special species and habitats	Genetic conservation	Conservation of genetic resource (Ennos et al. 2000) Use of local origins (Herbert et al. 1999)
	Special habitats	Restoration of native woodland on planted ancient woodland sites (Thompson et al. 2003) Creation of new native woodlands (Rodwell and Patterson 1994) Peatland restoration (Patterson and Anderson 2000)
	Special species	Red squirrel (Pepper and Patterson 1998) Forest birds (Currie and Elliott 1997) Raptors (McGrady et al. 1997, Petty 1998)
Habitat management and enhancement	Tree species diversity	Inclusion of broadleaves (Humphrey et al. 1998)
	Impacts of herbivores	Grazing (Mayle 1999, Gill 2000)
	Encouragement of structural diversity	Deadwood (Humphrey et al. 2002b); Silvicultural systems (Kerr 1999); edge management (Ferris and Carter 2000)
General	All aspects of biodiversity	Forest Nature Conservation Guidelines (Forestry Commission 1990)
	Site potential	Ecological site classification (Ray 2001)
	Forests and water	Forests and Water Guidelines (Forestry Commission 2000a)

conserve biodiversity are particularly effective and worthwhile (CJC Consulting 2003). The continuing low economic return from timber production has meant that profitability has reduced and that owners are likely to require incentives to afford biodiversity enhancement. The UK Forestry Standard (Anonymous 1998) provides the minimum requirement for access to the incentives provided by the grant schemes. A number of incentives have had a particular biodiversity focus – including challenge funds to restore coppice working (for the benefit of lepidoptera), to remove fencing where it was likely to cause mortality to flying woodland grouse, and to assist de-fragmentation of woodlands in agricultural landscapes. The latter scheme (JIGSAW) provides complete cost reimbursement but is awarded competitively. Grant schemes are currently being reviewed to develop a focus in line with country strategies (see below), but are likely to retain incentives to expand woodland area (particularly native woodlands), and to enhance existing woodlands through stewardship grants.

The UK Woodland Assurance Scheme (UKWAS) for certification provides an indirect form of incentive. The incentive is market access (e.g. with FSC logo) and achievement of a standard that is required and acceptable for formal incentive schemes. UKWAS sets out detailed guidance and requirements for managers seeking to meet the desired management standards, including a number of challenging criteria for biodiversity protection and enhancement (Anon 2000).

Changes in forest management

Management of UK forests has changed as a result of the changes in policy and delivery mechanisms. Some incentives have been notable for their success – for example new broadleaf planting (largely of native origin) has increased from less than 1000ha per year prior to 1980 to more than 11000ha/year in 2000; planting of new native pinewoods has increased from almost nil in 1990 to greater than 4500ha in 2000 (Rollinson 2003). Management of state forests has led the way, being a means of delivering policy through direct intervention. Forest Enterprise, the agency of the Forestry Commission responsible for their management has provided clear demonstration of biodiversity conservation and enhancement through protection of special sites and species, restoration of special habitats, development of strategic plans, and formation of partnerships with environmental groups. All Forest Enterprise forests have received UKWAS certification (Bills 2001). Significant habitat restoration has been achieved by a number of EU LIFE projects with a focus on Atlantic oakwoods, wet woodlands, the historic New Forest, bog restoration, and conservation management for the capercaillie.

3. An analysis of the recent policy developments – the latest policy agenda

Recent political developments have meant that the constituent countries within the UK have, for the first time, developed their own Forestry strategies (Forestry Commission 1998, 2000b, 2001) and separate biodiversity strategies (Scottish Executive Environment Group 2003). A summary of the biodiversity-related measures is contained in Table 4.

The similarities and distinctive emphases reflect both the political priorities but also the bio-physical characteristics of the regions. There is a general focus on lowland agricultural landscapes with small woods in the England strategy, and a greater emphasis on upland landscapes containing 20th Century planted forests in the Scottish and Welsh strategies. Despite this there are many common themes – including at the highest level the need to take on a wide range of other values and objectives in developing the overall forestry strategy. Biodiversity and the environment is only one of five or six main objectives.

Table 4. Analysis of biodiversity component of Country Forestry Strategies. Common over-riding objectives are forestry for - i. Rural Development; ii. Economic Regeneration; iii. Recreation, Access and Tourism; iv. Environment and Conservation; in addition for Scotland v. Creating a diverse forest resource, and for Wales vi. A new emphasis on woodland management

	England	Scotland	Wales
Main environmental objective	Forestry for the environment and conservation	Making a positive contribution to the environment	A diverse and healthy environment
Common themes	• Protection of existing woodland (adequacy of existing measures) and restoration of existing woodland (through management) • Expansion/extension of existing woodland, particularly where linking existing semi-natural woodland • Using woodlands to restore environmental qualities to post-industrial (or urban) landscapes • Integration of woodland with other land uses • Addressing and understanding the effects of pollution (atmospheric deposition leading to acidification and nitrification) • Developing the use of alternatives silvicultural systems to clearfell • Working in partnerships (with private sector, development agencies, local government, other government agencies and voluntary bodies) • Targeting grants (planting or stewardship/management) to specific topics, rare species, woodland types and geographic areas • Promotion of understanding of importance and values of (native) woods		
Distinctive emphasis	• Compensatory planting where woodland lost to development • Developing the use of long-term forest plans • Monitoring the biodiversity benefits achieved (through links to the National • Biodiversity Network) • Protecting archaeological sites from new planting	• Enhancing the diversity of farmland landscapes • Woodland grouse, and deer • Tackling acidification of lakes and rivers, if necessary through deforestation • Removal of non-native vegetation from ancient woodland sites • Restructuring of extensive plantations • Enhanced catchment planning – water management and reduction of flood risk	• Applying measures to enhance biodiversity in coniferous woodland • Conserving Welsh landscapes, including historic parks • Incorporating (and restoring where appropriate) non-woodland habitats such as heath and bog
'Ecological' Assumptions	• De-fragmentation is good • Local seed sources for trees and shrubs are good • Biodiversity Action Plans are a good guide	• Mixed species forests are good • Some grazing is beneficial • Restructuring is good • Riparian woodlands are good • Restructuring is good • Certification is good	• Increase in core area of native woodland habitats is good • Continuous cover (multi-aged structures) is good for biodiversity

There is clearly a strong emphasis on *protection* of existing woodland – consistent with the small area of remaining semi-natural woodland. The focus of the protection differs between countries – with problems of development, neglect, and need for management all being identified. Restoration of these habitats is encouraged.

A further strong emphasis is on *expansion* – but with different underlying reasons, including replacement of any further losses, additions to landscape diversity, linkages to remnant areas giving benefits of de-fragmentation, and as a form of post-industrial habitat restoration.

There is also a greater focus on management at the landscape scale with recognition that there should be better *integration* of forestry and other land uses. Agriculture is a clear target, but so are the hydrological interests within catchments – both for water quality and also downstream flood control. The intention is to maintain a range of qualities and services – including visual aesthetics.

Management of *existing extensive forests* receives relatively less attention. The two exceptions are aims for an increase in restructuring, and the application of silvicultural systems to replace clear-felling. The need for restoration of some other habitats within these extensive forests is also noted. Interestingly, only one country (Wales) aspires to enhance the biodiversity value of planted coniferous forests and assumes that this can be achieved by practising continuous cover forestry. The need to control deer, and to improve conditions for woodland grouse are highlighted in the Scottish strategy.

As highlighted earlier in the paper, the next steps in delivering policy should be to form an evidence base from which to monitor, set standards and provide suitable controls and incentives. Further development of the English Forestry Strategy has highlighted improving the evidence base as one of the six priority actions (Forestry Commission 2002a). The country strategies contain little explicit statement of research needs, though it is accepted that identification of gaps in knowledge and understanding should be a next step. In due course, the identification of knowledge gaps will help to shape a new UK forest research strategy.

4. Preliminary analysis of the knowledge gaps in responding to the new policy developments

The following is a preliminary analysis of possible research implied by the strategies. This has been undertaken within the context of contributing to the development of European research networks, and does not represent any form of official response to the strategies or precedent for a domestic research strategy. The following merit preliminary attention and comment:

Are there features of the strategy that are unsupported by research findings?
Yes, many are currently unsupported. Many of the ecological assumptions (see table) are based on very broad inferences from the literature (often international) with little supporting evidence from the UK. This is unsurprising given the rapid (yet welcome) progress in policy, and is by no means unrepresentative of the situation elsewhere.

Does implementation of policy require new research?
Yes, implementation of policy will require the identification of a range of accompanying measures, and whether sufficient evidence exists to specify these. It is clear that the knowledge base is inadequate, and much has to be progressed on the basis of heroic assumptions. For example, it is generally agreed that proximity to existing woodland habitat

enhances the benefits to be gained from new planting – but how can this be specified in a way that encourages planting in the right place? The Scottish Forestry Grant Scheme has adopted a 300m buffer rule as a way of targetting the incentive but the ecological underpinning for this threshold is unclear. Further work could refine this threshold to take account of species specific characteristics, and the landscape context, thereby avoiding unwitting homogenization through the application of rules.

Has recent research identified aspects that are apparently under-represented?
Yes, there appears to be a lack of consideration of the merits of the large exotic forests, and yet recent findings have highlighted the benefits of managing these with biodiversity in mind (Humphrey et al. 2002a, Kanowski 2003). Others have identified the strategic benefits of the ecologically wise management of the bulk of forests, provided that a natural reserve structure is also in place (Hunter Jr 1999). The lack of attention may be because the existing UK Forestry Standard is held to deal with the key points, but there are dangers that the lack of profile leads to an undervaluation of the resource. Research has indicated that a characterisation of exotics as bad and natives as good is incorrect, and yet this is an implicit theme in much policy-related discussion with respect to biodiversity.

What appear to be the specific gaps in knowledge required to support delivery?
The following appear to be some of the larger gaps in knowledge and understanding needed to deliver policy:

- How can the acceptable balance of different land uses be determined? And how can this be achieved, especially when there are multiple ownerships, and mix of managed and unmanaged land?
- What scale of remnant habitat is valuable and therefore worth enhancing? How does this relate to edge conditions, and the resilience of the habitat to further external forces such as climate change, pesticide drift, and atmospheric drift?
- What is an appropriate rate of restoration? Is there a trade-off between removing exotics, but in doing so also removing woodland structure that may be creating suitable (albeit temporary) micro-habitat conditions?
- How soon will the biodiversity benefits evolve in new planting, and what can be done to enhance the process? What are realistic time scales for restoration and establishment of new habitats? How important are silvicultural systems in influencing the biodiversity of these forests?
- Are old growth and core area specifications developed in forest-rich regions relevant to countries with highly modified forests and low proportions in landscapes? Do the necessary species guilds exist to exploit such habitats? Which species, if any, should be re-introduced and how is this best achieved?

In addition, there are a large number of specific questions linked to the tactical and operational implementation of policy – for example identifying appropriate densities of deer populations, controlling alien species such as the North American grey squirrel (*Sciurus carolinensis*) and Rhododendron (*R. ponticum*).

How can these knowledge gaps be filled?
The increased spatial and temporal scale, and the novel combination of scenarios places new demands on researchers. There is a need for new methods of investigation. Many of the knowledge gaps are not susceptible to traditional site-based research. Many of them concern scenarios where the combination of conditions does not currently exist to be investigated. It is clear that there will have to be a much greater reliance on models and modelling frameworks.

The models must have appropriate data sources, be explicit about their assumptions, and make clear the degree of uncertainty. The frameworks must begin to provide an integration across scales and across disciplines. For example, biodiversity is often only one of a number of objectives that are required in participatory planning tools.

In addition, it has been suggested that **transference of knowledge** is also a suitable tactic. How can knowledge be transferred from one geographic location/habitat type to provide valid evidence? Some have suggested a need for a more systematic approach to review, akin to the process commonly used in medicine (Pullin and Knight 2001). But is sufficient known and documented about the 'environment' in which studies have been performed, and are the studies occurring at the relevant spatial scales? An example of this conundrum concerns maintenance of large core areas of old growth forests. This is an inherently attractive proposition, whose value has been established in extensive forest landscapes in North America and, to a lesser degree, Scandinavia and Australia. The proposition has been widely adopted in policy and in standards – but are these sensible aspirations in highly managed and fragmented landscapes? Transference of approaches is not always successful. For examples, we encountered difficulties in applying a forest landscape analysis approach (Diaz and Apostol 1992) to British landscapes, particularly those with a significant cultural heritage (Bell 2003), and in adopting practices of mimicking natural disturbance (Quine et al. 1999).

4. Concluding remarks

Much recent development in policy and practice has been very good for biodiversity in UK forests. There is strong protection for much of the remnant semi-natural woodland habitat, and for special species. There are substantial commitments to enhancing conditions in a range of existing and new (particularly native) woodlands. However, there are some very difficult questions implicit in the new strategies. Some of these concern the integration of biodiversity and environmental benefits with other aspects of sustainable forest and landscape management. Others concern specific knowledge required to support strategy and assist in delivery of policy. There is a general need for the evidence base to catch up (and get ahead of) policy and practice, and conversely for policy to incorporate some recent findings. However, further discussion is required between policy makers, practitioners, researchers and public representatives to prioritise effort both domestically and within the broader European area.

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The Contribution of Structural Elements to Plant Diversity in Mediterranean Forest Landscapes

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Abstract

Sustainable management and maintenance of biodiversity at the landscape level demands methods of biodiversity evaluation that take into account all the structural elements of a landscape (matrix, habitat cores, habitat edges and linear features). Fragmentation is a growing feature of the Mediterranean rural landscapes and it results in the reduction of habitat core areas and proliferation of edges or boundaries between different land-uses. New boundaries and linear features, such as roadsides, riversides or hedgerows, are important as refuges for some taxa and must also be considered as functional elements in the landscape mosaic. In this paper, we propose an improved method for assessing and monitoring vascular plant diversity in the Mediterranean rural landscape taking into account these structural elements mentioned above. Using the municipality as the landscape unit, land-uses were defined according the CORINE land cover classification, based on interpretation of recent aerial photographs, and cross-referenced to EUNIS habitat classification. Each habitat type was stratified and core habitats were sampled together with a number of linear elements, randomly selected among the more frequent types. The municipality of Cadalso de los Vidrios was chosen as the landscape unit to sample in this study in order to test methodology and obtain some preliminary conclusions. The sampling unit for recording the presence and abundance of vascular plant species was a multi-scale plot, measuring 20 m x 50 m (1000 m^2). Non-parametric and parametric methods were used to evaluate sampling efficiency in order to identify the most accurate method. Randomised accumulation curves were calculated to estimate species richness, and non-weighted additive models were used to estimate the Shannon diversity index at the landscape scale. Plant species richness was significantly

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underestimated when linear elements of the landscape were not included in the estimation. Finally, we propose the use of some landscape metrics for the weight contribution of structural elements to landscape diversity.

Keywords: Biodiversity estimators; patch core; patch edge; Shannon index; species richness; Mediterranean landscape

1. Introduction

The maintenance or enhancement of biodiversity must always be included as an essential part of assessing sustainable forest management, so efficient methodologies must be developed for the evaluation of biodiversity. The landscape scale, studied by remote sensing techniques, is the most valuable referral for the analysis of land structure and evolution variables, as has been shown by many studies carried out since the eighties (Turner and Gardner 1991).

From this point of view, a landscape is considered as a system of many structural elements including its matrix, core habitats, edge habitats (borders or areas of interaction between habitats) and linear features such as roadsides, riversides or hedgerows. Various metrics are available for analysing landscape structure, such as patch density, patch core area, patch edge length, and the spatial distribution of patches like contagion, proximity, connectivity, interspersion, fragmentation, evenness and patch diversity (e.g. Forman 1995; Hansson et al. 1995; McGarigal and Marks 1994).

Many studies have already shown the importance of certain structural metrics for the conservation of different species. Moreover, many of these metrics are correlated when they are applied to the landscape as a whole (Riitters et al. 1995) and also at the class level of patches (Bolaños et al. 2001).

To assess biodiversity at habitat and landscape levels, variables that can be easily analysed and correlated with ecosystem conservation are needed. The composition of plant communities has long been used to develop biological indicators (Mueller-Dombois and Ellenberg 1974). More recently, plant communities have been proposed as a way of describing biodiversity at a landscape level (Noss 1987) and they have been tested as variables to discriminate and classify ecosystems using remote sensing technology (Treitz et al. 1992; Schriever and Congalton 1993; Franklin et al. 1994; Wolter et al. 1995; Nagendra and Gadgil 1999a: 1999b).

At the landscape level, it is possible to study the composition and distribution of the vascular plants that determine forest biodiversity, and the relationships established with neighbouring land use types that could be considered as species reserves. This is particularly important in areas of forest exploitation, where biodiversity may temporarily reach minimum values.

For this integrated landscape analysis we propose a methodology involving: (1) remote sensing information, to identify land cover and land use types; (2) analysis of environmental factors such altitude, aspect, and cover density, to discriminate the composition of habitats in each land cover type; (3) multi-scale field sampling techniques, to asses plant diversity; (4) affinity analysis of plant community composition such as Scheiner's Analysis (1992), to validate the stratified random sampling design, and (5) the additive model that partitions gamma diversity into its alpha and beta elements.

In order to assess habitat biodiversity at the landscape level it is necessary to weight the plant communities found in the different landscape elements by the metrics of their structural components. Our team has tested this methodology in three Spanish forest landscapes following a sampling design in which patch habitat cores were the only landscape structural elements. In that first approach, the best weight function in segregating municipalities was the number of patches multiplied by IJI (Juxtaposition and Interspersion Index) (Ortega et al. 2004). IJI is a measurement of the spatial distribution of landscape patches that is based on the patch edge length and the type of contact between neighbouring patches.

Habitat fragmentation in the north Mediterranean countries resulting from increased population density and land communication networks has resulted in the increasing influence of linear structural elements in the assessment of vascular plant diversity at the local level. This is well known from studies of animal communities, because many linear elements establish natural biotope boundaries and natural limits of dispersal areas for some species, as well as corridors that link different parts of habitats that support metapopulations (Anderson and Danielson 1997). In plant communities, this influence seems to be positive when the number of species is higher in the edge habitat than in the core habitat, as reported in a small forest in Northern Belgium (Honnay et al. 1999).

To assess the influence of other landscape structural elements, we are at present working on five Mediterranean rural landscapes We present here the results obtained on one of those five landscapes: the one which shows the highest number of different linear elements and core habitat types, and has more than 50% of its area covered by forest.

The aim of this paper is to analyse the influence of linear structural elements on the estimation of local vascular plant richness and the Shannon diversity index. The linear elements considered are habitat boundaries (forest edges and grassland edges) and linear features such as hedgerows, roadsides, and riversides. We compare local species richness and diversity values of the Cadalso de los Vidrios landscape computed by statistical estimators based on two sets of sampling plot: i) all plots combined (core habitats together with linear elements plots) and ii) plots of core habitat only.

2. Materials and methods

2.1 Study area

The municipality of Cadalso de los Vidrios (Madrid) is located in a homogeneous Spanish biogeoclimatic zone as described by Elena-Roselló (1997), and is a good example of a meso-Mediterranean forest landscape.

The area lies on the southern foothills of the ‘Sierra de Guadarrama’. Geologically, the landscape is dominated by granite outcrops on a gentle slope of arcosic rocks of variable resistance to erosion. The total area is 4762 ha, and rises from an altitude of 601 m to its highest point at 1209 m a.s.l.

The population of Cadalso de los Vidrios has grown during the last decade due the proximity of Madrid city. New urban developments of second homes have been built in former pine forest areas. Traditional land use for agricultural crops (mainly vineyard) and grazing are in sharp decline, but quarrying of the granite outcrops has grown during recent years. Phytosociologically potential vegetation is *Junipero oxicedri – Querceto rotundifoliae sigmteum* communities, and *Luzulo fosteri – Querceto pyrenaicae sygmetum* in the highest areas (Rivas-Martínez 1987).

The main habitat types are as follows: coniferous forests (*Pinus pinea* and *P. pinaster*) = 44%; broadleaved forests (*Quercus ilex*, *Q. pyrenaica* and *Castanea sativa*) = 11%; land principally occupied by agriculture, with significant areas of natural vegetation = 27%; sclerophyllous vegetation = 8% and grassland = 2%. Also, because it is a rare habitat type, we mention the barren areas with litho sols and granite boulders.

Table 1. Habitat cores and linear elements sampled in the municipality of Cadalso de los Vidrios with level 3 Land Cover CORINE classification cross referenced to EUNIS habitat classification.

Type of habitat	Land cover CORINE	EUNIS habitat classification	Plots
Cores	Natural Grassland	Mediterranean Xeric grassland	2
	Sclerophyllous vegetation	Arborescent matorral	1
	Coniferous forest	Lowland to montane Mediterranean *Pinus* woodland [*P. pinea* and *P. pinaster*]	2
	Broadleaved forest	Mediterranean evergreen woodland [*Quercus ilex*], Thermophilous deciduous woodland [*Quercus pyrenaica*] and [*Castanea sativa*]	3
	Bare rock	Temperate-montane acid siliceous screes	3
	Green urban areas	(*)	1
Linear elements	Road and rail networks and associated land	Weed communities of transport networks and other constructed hard-surfaced areas	1
	Water courses	Acid oligotrophic vegetation of spring brooks	1
	Land principally occupied by agriculture, with significant areas of natural vegetation	Rural mosaics, consisting of woods, hedges, pastures and crops	1
	Natural Grassland edge	Mediterranean Xeric grassland	1
	Coniferous forest edge	Lowland to montane Mediterranean Pinus woodland [*P. pinea* and *pinaster*]	1
	Broadleaved forest edge	Thermophilous deciduous woodland [*Quercus pyrenaica*]	1

(*) No cross-reference exists between CORINE and EUNIS classification. The habitat is lowland to montane Mediterranean *Pinus* woodland [*P. pinea and P. pinaster*] with built-up areas.

2.2 Habitat definition and delineation of landscape structure

This was carried out after interpreting aerial photographs (1998, MAPA aerial survey for the Olive inventory) and the subsequent definition of thematic layers of land use and vegetation cover using 3rd level CORINE land cover typology cross-referenced to the EUNIS habitat classification (EEA and ETC 1999). After delineating habitats (Table 1), a stratified sampling strategy was implemented for selecting habitat core plots based on aspect as the only criterion. Elevation range is not great enough to be a factor in determining vegetation types, but aspect is expected to influence plant composition and diversity (Bales et al. 1998; Miller and Franklin 2002). Two aspect categories (South-exposure from 90 to 270º referred to as "sunny", and North-exposure from >270 to <90º referred to as "shady") were identified from a Digital Terrain Model (DTM) and they were sampled in each core habitat. A small number of linear structural elements were randomly selected among the more frequent habitat types by overlaying the area of cover of each habitat with communication lines and water networks using GIS (ArcGis 8.2). A hedgerow was sampled for the habitat type “Land principally occupied by agriculture, with significant areas of natural vegetation” (Table 1).

2.3 Vascular plant sampling and estimates of species richness and diversity

Sampling of vascular plant species was carried out in multi-scale plots using 0.1 ha sample areas (50m x 20m) as suggested by Stohlgren (1995) modified from Whittaker.

Subsampling followed the same methodology as used by Ortega et al., (2004) with 10 plots of 2 m x 0.5 m (1 m^2), two of 2 m x 5 m (10 m^2), and one of 5 m x 20 m (100 m^2). Within these subplots the abundance of herbaceous and woody plants was estimated in 5 cover categories: 1) <12% of total subplot area; 2) between 12 and 25%; 3) between 25 and 50%; 4) between 50 and 75%, and 5) >75% Plots of 1000 m^2 were fully surveyed for new species not found in the 1, 10 and 100m^2 subplots. An abundance rate of 0.1% was assigned to any of the new species found in the 1000 m^2 plots,

To record major environmental variability, each habitat core plot is located 100 meters from the nearest edge following the maximum slope direction. The linear element plots have their longest boundary perpendicular to the habitat borderline or linear feature and 25 meters inside each neighbouring habitat. Fieldwork was carried out in May and June 2001 and 2002, the most appropriate phenological season for the identification of plant species.

2.4 Data analysis

Species richness was estimated using the following statistics:

Non-parametric estimators:
1. First-order incidence-based Jack-knife estimator (Jack1), developed by Burnham and Overton (1978: 1979). Based on the number of unique species (species found only in one plot), and the number of plots sampled.
2. Abundance-based estimator (Chao1), developed by Chao (1984). Based on the number of singletons (species represented by a single individual).
3. Incidence-based Coverage Estimator (ICE), is based on species found in 25% (or fewer) sampling units.
4. Abundance–based Coverage Estimator (ACE) is based on those species with a relative abundance of 0.1% (or fewer) in the sample.

ACE and ICE are modifications of the Chao and Lee non-parametric estimators discussed by Colwell and Coddington (1994).

Parametric estimators:
1. Logarithmic equation.
2. Exponential equation.
3. Clench equation. Developed by Clench (1979) as a modified Michaelis-Menten equation.

EstimateS software (Colwell, 1997) was used to draw cumulative randomised curves for species richness and the Shannon diversity index. Collector’s curves or maximised species accumulation curves were drawn by selecting the plot inventory with the “highest possible number of new species”. The plot of the largest species forces the curve starting point.

The input of new species from the plots located in the linear landscape elements was analysed by comparing the curves with all the sampled plot data and those using only the habitat core data.

3. Results and discussion

3.1 Sampled plot characteristics

18 multi-scale plots were sampled in Cadalso de los Vidrios (12 habitat core plots and 6 linear element plots). Mean number of species per plot was 75.6 and mean number of species unique to the plot was 6.2. Both variables were significantly higher (F=9.10; p=0.008 and F6.55; p=0.02, respectively) in linear element plots (mean= 91.8 species and 8.8 unique species) than in habitat core plots (mean = 67.4 species and 3.5 unique species). The Shannon alpha diversity index was 2.7 with no significant differences between linear element plots and habitat core plots.

3.2 Estimate of local plant species richness

The cumulative randomised curve of species richness computed from habitat core plots only, showed significant differences from the curve of all the sampled plots combined (Figure 1). The comparison is made by ascertaining whether the diversity of habitat cores lies within the 95% confidence limits of the rarefaction curve of the total rarefied curves. The same kind of curve computed using Chao1 and ACE species richness estimators is not significantly different from the curve computed with all sampling plots combined.

The ACE estimator is characterised by a significant approach to the randomised curves of the observed data when the species show random distribution among the samples (Chazdon et al. 1998). This effect is common when the communities are especially rich in vascular plant species, as occurs in the habitats of the landscape under study.

The relationship of Chao 1 and ACE estimator to the observed data is not conclusive and therefore, neither is their value as an estimator. Sampling deficiencies are possible due to the presence of annual ephemeral species that do not germinate every year due to unsuitable climatic conditions (Peco 1989), or do not germinate in spring time, or are misidentified. It seems that ICE and Jack1 estimators, that predict 20% and 26% more species, respectively (Figure 1), could be closer to reality; ICE being the one that shows a more asymptotic behaviour. Bias for this estimator has been found to be smaller than for other non-parametric estimators for different types of species abundance distributions and spatial distributions of species (Chazdon et al. 1998) and it is recommended when the sample contains at least 80% of the total number of species in all samples combined (Wagner and Wildi 2002). ICE also overestimates the species richness of other types of communities such as the arthropod species (Finnamore et al. 2002)

In an attempt to analyse sampling efficiency, the collector's curve or maximized species accumulation curves (*Cmax*) computed from all sampled plots combined as well as the one computed using core habitat plots only, were fitted to exponential, logarithmic and Clench (Clench 1979; Soberón and Llorente 1993) equations. The best results were obtained with the Clench´s asymptotic equation (Table 2)

Species richness values predicted by the Clench´s curve asymptote using all the landscape elements are 50 species higher than the observed values, which indicates a sampling efficiency of 87.4% . When we use only patch core plot data, the predicted species richness is lower than that of all the landscape elements, reducing the efficiency to 80.7% . The logarithmic equation has the poorest fit to the observed value curve. The exponential equations show lower predictions than the richness observed for all the sampling plots combined, but they fit exactly when we consider the patch core plot data alone.

3.3 Estimation of local vascular plant diversity

Randomised cumulative curves of the Shannon diversity index calculated from all plots combined and from habitat core plots only, show a more asymptotic curve than the species richness (Figure 2). Using only 12 habitat core plots, we found significant differences in diversity values from the observed data in 12 core and 6 linear elements plots. We found significant differences when predicting species richness from 9 plots where there are fewer species in habitat core plots than in core and linear elements plots (Figure 1).

According to Lande (1996), the Shannon index randomised curves are more asymptotic than species richness curves, so the difference between core habitat plot sampling and core and linear element plot sampling is reduced. Local diversity calculated using the Shannon index, with the data from all the different habitat types existing in the landscape, seems to be only slightly sensitive to the sampled habitat type: patch core habitat or linear element. Linear elements provide significantly more species, but the species abundance distribution in the local pool shown by the Shannon index, seems to be stable. These trends should be studied in other landscape types in order to reach sound conclusions about the importance of linear elements when investigating richness and diversity of vascular plant species at a local scale.

The estimation of vascular plant diversity and richness including both core habitat and linear elements as structural elements of landscape on a local scale seems to be an useful approach to monitoring the local species pool in a given area. The stratified sampling design used is based on the selection of a sample plot in each type of habitat core in relation to the main environmental factor, as proposed by Wagner and Wildi (2002), and plots representing the main linear structural element types existing in the landscape. Together with a sample plot design adequate for different vegetation types (Stohlgren et al. 1995) it allowed us to reach a good approximation to the local species richness and diversity of the studied landscape.

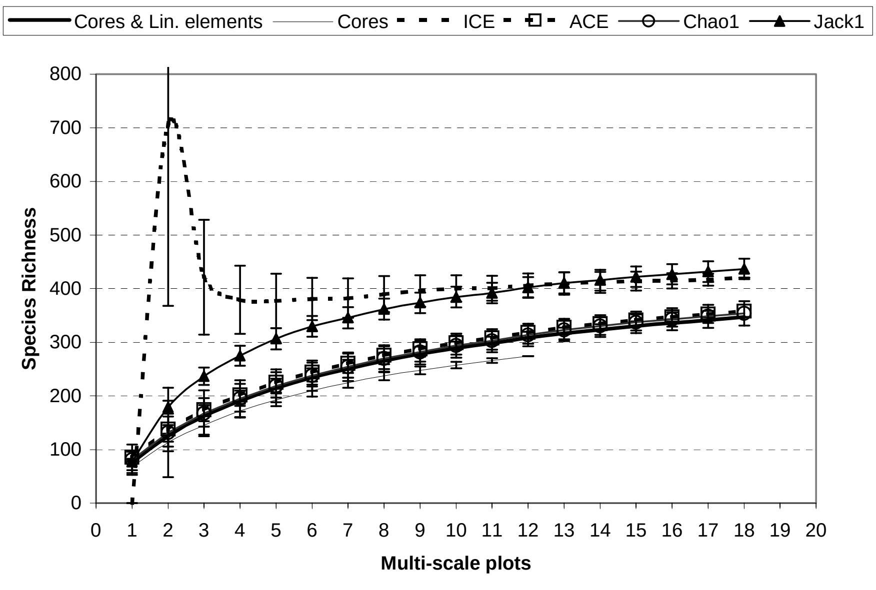

Figure 1. Cumulative randomised curves of all the 0.1 ha plots recorded in Cadalso de Los Vidrios. The species richness curves predicted by ICE, ACE, Jack1 y Chao1 are shown. Data are indicated with their standard error.

Table 2. Curve fittings of maximum accumulated species curve to the Clench (1979), logarithmic and exponential equations.

Type of equation	Equations	Final residues	Explain variance (%)	Asymptote (species)	Observed Species richness	Sampling effort (Plots)	Efficacy of sampling (%)
	Cores and linear elements						
Clench	y=(166.78)*x/(1+(0.42)*x)	318.5	99.56	397	347	18	87.38
Logarytmic	y=(132.93)+((80.12)*log(x))	1795.12	97.51	-	347	18	
Exponential	y=(114.32)/(0.33)*(1-exp(-(0.33)*x))	1156.75	98.39	341	347	18	
	Cores						
Clench	y=(138.74)*x/(1+(0.41)*x)	315.5	99.14	338	274	12	80.68
Logarithmic	y=(100.52)+((75.38)*log(x))	909.55	97.52	-	274	12	
Exponential	y=(104.84)/(0.38)*(1-exp(-(0.38)*x))	24.21	99.93	274	274	12	

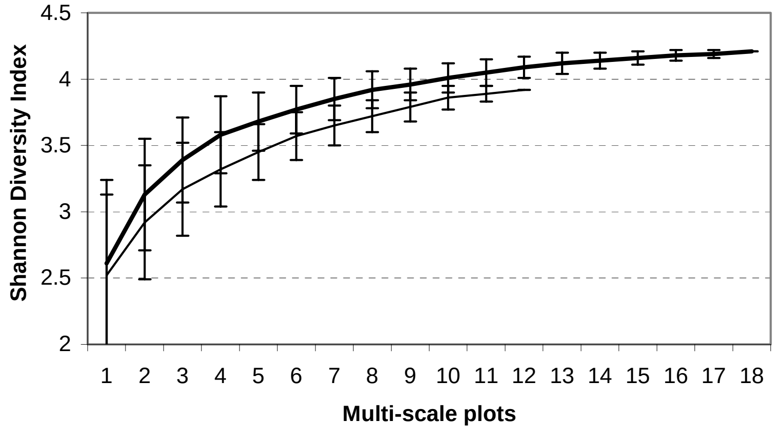

Figure 2. Randomised curves of the Shannon diversity index increment calculated with all the 0.1 ha sampled plot (thick line) and with only 0.1 ha core patch plot (thin line). Values are shown with standard errors.

Nevertheless, weighting each habitat diversity according to the landscape metrics related to their structure (such as total area, number of patches or spatial distribution of both patch types), would allow us to differentiate land both spatially and temporally. Approaches like this one, linking biodiversity assessment with landscape structure are not only useful, but necessary in the framework of sustainable rural development.

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Maintaining Biodiversity in Managed Forests – Results of Beetle and Polypore Studies in Boreal Forests

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Abstract

This paper gives a review of several studies where the effects of different forest management methods on polyporous fungi and beetles, and the importance of woodland key habitats (WKHs) for polypore diversity were investigated in different zones of boreal forest in Finland. The total species richness of beetles was relatively well maintained in the seed-tree cut pine forests over the time span of 15 years, but the proportion of rare species had decreased, obviously because the lack of suitable substrate. In the clear-cut spruce forests which were planted for pine the beetle community had changed considerably due to the change of the major tree species and changes in microclimate. Proportion of rare species was low on clear-cuts. The polypore diversity in the seed-tree cut pine forests was relatively well maintained on the pre-logging CWD still 40 years after logging, but on the logging waste only about half of the total species richness was found. In the selectively logged spruce forests neither the forest structure nor the diversity of polypores had returned to natural level in 60–100 years. The results show that there is a time lag of several decades before the real effects of logging are detectable in the saproxylic and saprotrophic communities. WKHs selected on the basis of rich vascular plant flora maintained poorly the characteristic polypore flora of boreal forests. Small (0.2 ha) WKHs covered only about 20% the detected total species richness of polypores. Leaving large-diameter retention trees and CWD in regeneration areas would benefit many saproxylics. It seems, however, difficult to maintain in managed forests the diversity of those saproxylics, which have adapted to the steady, moist microclimate of old-growth spruce forests. Relatively large (0.5–1 ha) unlogged patches of typical old-growth forests would help to maintain both polypore and beetle diversity of spruce stands.

Keywords: biodiversity; beetles; polypores; boreal forests; woodland key habitats.

1. Introduction

Fennoscandian countries are often mentioned as model countries for sustainable forestry. When considering timber production only, the statement is undoubtedly true (Finnish Forest Research Institute 2002; Skogsstyrelsen 2002), but not if we consider the ecological sustainability of forestry, including biological diversity of species. Within the last 100–150 years the species diversity of forest ecosystems in Fennoscandia has declined considerably, and the number of threatened forest species is high in all the Nordic countries which implement intensive forestry (Direktoratet for Naturforvaltning 1998; Gärdenfors 2000; Rassi et al. 2001). In Finland, for instance, forestry is the primary reason of threat for 34.7% of all the threatened species (Rassi et al. 2001).

The main changes in the forest structure affecting the species diversity are decrease in the volume of coarse woody debris (CWD), decrease in the area of old-growth forests, decrease in the amount of deciduous trees and fragmentation of forests (Esseen et al. 1997; Rassi et al. 2001). In Finland, decrease in the amount of CWD is considered as the main single threat for forest species (Rassi et al. 2001). Saproxylic species (species which are directly or indirectly depending on decomposing wood) comprise a large portion of the biological diversity of boreal forest ecosystem, including a large number of macro- and microfungi, invertebrates, lichens, bryphytes and some vertebrates (Samuelsson et al. 1994; Siitonen 2001). Siitonen (2001) has estimated that of the total species number in Finland (42 000–50 000 species) about 45% are forest species, and of those, about 20–25% are saproxylics (4000–5000 species). Beetles (*Coleoptera*) and polyporous fungi (*Basidiomycota*) are among species groups which have been affected by CWD decline. Approximately 40–67% of forest beetles in Fennoscandia are saproxylics (Stokland 1994; Martikainen et al. 2000; Sippola et al. 2002). Respectively, the majority of polypores are wood-decomposing organisms being directly dependent of CWD as their substrate (Gilbertson and Ryvarden 1986).

Several factors affect the survival of saproxylic species in forest ecosystems. The main factors that are connected with forest structure and dynamics are substrate availability, microclimatic conditions, dispersal possibilities and disturbances (Speight 1989; Samuelsson et al. 1994). Because CWD is a temporary substrate which decomposes in the course of time, both spatial and temporal availability of new suitable substrate has to be secured (Speight 1989). A large number of saproxylic beetles, including many rare and threatened species, are adapted to young successional stages or sun-exposed sites created after natural or human disturbances (Jonsell et al. 1998; Martikainen 2001; Similä et al. 2002 a; Lindhe 2004). Also many polypores can inhabit open and sun-exposed stands (Sippola and Renvall 1999; Lindhe 2004). On the other hand, many old-growth forest polypores which inhabit large-diameter trunks require steady, moist habitat (Renvall 1995; Kotiranta and Niemelä 1996), and also many old-growth forest beetles which are confined to the sites of long forest continuity survive only in closed forests with steady microclimatic conditions (Mannerkoski 1996). Poor dispersal ability of a species may restrict the colonization of new substrate in heavily fragmented environment, and, in the course of time, either cause local extinction or deteriorate the genetic structure of a population. It has been detected that in polypores fragmentation, local and regional amount and age of source areas as well as geographical location of a species in its distribution area affect the spore deposition and spore viability (Högberg 1998; Edman et al. 2004 a; b).

Intensive forestry has influenced the forest structure of Fennoscandia for 100–150 years, but the changes in the species diversity have been observed mainly during the last few decades (Esseen et al. 1992; 1997). Several new practices are now implemented in practical forestry to mitigate the effects of forestry. Coarse woody debris, retention trees and woodland key habitats (WKHs) are left in the logging areas to maintain diversity, and corridors and stepping-stones are preserved to diminish the effects of fragmentation (Forest and Park Service 1998; Meriluoto and Soininen 2002). Undoubtedly, all these measures will enhance species survival in forestry areas. However, we still need a lot of knowledge about the efficiency of these practices, especially in relation to different organism groups. This article gives an overview of the results of several studies, which aim to investigate how different forest management methods influence the species diversity of beetles and polyporous fungi, and how WKHs selected by rich vascular plant flora are able to maintain species diversity of polypores.

2. Materials and methods

The beetle studies were conducted in 1992–1993 in Finnish Lapland, in the nothern boreal forest zone near the timberline. Beetles were trapped through the whole summer on one-hectare size study sites using free-hanging window flight traps. In 1992, the number of traps per study site varied between one and three, and in 1993 five traps per each study site were used. The study sites represented different forest site types and treatments as follows (number of replicates in parethesis): 1) Old-growth pine forests (15) 2) Old-growth spruce forests (6), 3) 1-year old seed-tree cut sites of pine forests (3), 15-year old seed-tree cut sites of pine forests (3), 4) 15-year old clear-cuts of spruce forests, with soil treatment by ploughing, planted with pine seedlings (3). On recent seed-tree cut sites, 80–120 seed-trees per hectare were left to the sites to ensure regeneration. Seed-trees are removed from where saplings are over two meters high. On the 15-years old sites the mean height of saplings was 3.5 m, and the seed-trees had been removed. Besides of beetle trapping, living trees and CWD were measured at the sites. All CWD with >10 cm base diameter or 1 m length were measured by tree species, position (standing dead trees, logs, snags, branches, stumps, logging waste) and decay stage (range 1–5, from fresh to completely soft CWD). Detailed descriptions of the study areas and methods are given in Sippola et al. 1995; 1998 and 2002.

Wood-decomposing fungi were studied in 1996 in the old-growth pine forests and seed-tree cut areas using the same study sites as above, but older areas, logged with seed-tree cutting about 40 years earlier, were also included in the study. The numbers of replicates were: 1) Old-growth pine forests (4), 2) 3-year old seed-tree cut areas (2), 3) 18-year old seed tree cut areas (2), 4) 43-year old seed-tree cut areas, where the saplings were on average 7.4 m high (2). On each study site, fruit bodies of wood-decomposing fungi were inventoried on rectangular sample plots with an area of 3000 m^2 each (for details, see Sippola and Renvall 1999).

The effects of old selective loggings, conducted 60–100 years ago, were studied in nothern Kainuu, at the border of north and middle boreal forest zones. The areas were spruce forests, and the logging had been directed to the largest trees of the stands. In this study, the numbers of replicates both in natural and selectively logged forests were five, and on each study site polypores were inventoried on eight circular sample plots with a radius of 10 meters. Inventories were made in two successive summers in 1997–1998 (see Sippola et al. 2001 for details).The response variables in the beetle study were the total species richness, species richness of saproxylics and the number and abundance of rare species (at most 25 records from Finland in 1960–1990, Rassi 1993). In polypore studies, total species richness and the number and abundance of indicator species of old-growth and virgin forests, and the number of threatened species (according to Kotiranta and Niemelä 1996) were used as response variables. In addition, we separated species growing on coniferous and deciduous CWD as well as early, mid and late decomposers (according to Renvall 1995). The explanatory environmental variables included volumes of different CWD qualities and living tree volumes by tree species. In beetle studies, which included different forest site types, additional site variables such as number of vascular plant species, the coverage of shrubs and depth of humus layer were measured.

Rarefaction (Simberloff 1978) was used to compare species richness among different samples sizes, and Mann-Whitney U-test and Kruskall-Wallis one-way ANOVA were used to test differences between expected species numbers for given sample sizes. Variation in the species composition among stand and treatment categories were examined by correspondence analysis (Ter Braak & Šmilauer 1998) and Sørensen and Renkonen similarity indices (Spellerberg 1991, Renkonen 1938). Correlations between site variables and species richnesses were examined by Spearman rank correlation analysis (for detailed descriptions of statistical methods, see the original papers).

Preliminary results of a study on the importance of woodland key habitats for polypore diversity are also given in this paper. In Koli National Park, in the southern boreal zone, the polypore diversity of spruce-dominated WKHs was compared with the diversity in typical old-growth spruce forests surrounding the WKHs. Both the control sites (5 replicates) and the WKHs (15 replicates) had equal canopy density of spruce, but the WKHs were characterised by a rich vascular plant flora. Because the sites were located in the national park, all the areas were unlogged. Polypores were inventoried on circular sample plots with a radius of 10 meters; the number of plots varied 1–8 per study site. Resampling was used to detect differences in species richness between the originally different sample sizes. From both WKH and control data, 1, 5, 10, 15, 20, 25 and 30 study plots were chosen randomly with replacement. Observed species richness and abundance was calculated for each resample, and the procedure was replied 1000 times. The cumulative species number as a function of increased plot number was detected from the resampling data, as well as the cumulative number of species that was common to both WKH and control plots. The differences in the numbers of species growing on different substrates were tested by Mann-Whitney non-parametric test, these tests were performed with data per each study plot (Sippola et al. 2004).

3. Results

3.1 Effects of seed-tree and clear-cutting on beetle diversity

In the northern boreal zone, pine forests are naturally relatively open and sparse. On our study sites, the volumes of living trees in old-growth forests varied from 42 to 145 m^3 ha^{-1}. The volume of CWD varied from 1.4 to 39.8, being on average 18.8 m^3 ha^{-1} (Sippola et al. 1998). The large variation in the volume of living trees reflects differences in the nutrient content, altitude level and exposure between the stands, whereas the variation in the volume of CWD reflects the volume of living timber, past disturbances and decay rate (Harmon et al. 1986; Sippola et al. 1998). The volume of CWD increased in seed-tree cutting areas immediately after logging, being on average 25.7 m^3 ha^{-1}. The increase is due to the logging waste, which comprises branches, tree-tops, stumps and rotten pieces of trunk, which are not usable for pulp or construction timber. In the 15-year old logging areas the smallest pieces of CWD had mainly decomposed, and the volume of CWD was on the average at the same level than in the old-growth forests (Table 1). The accumulation of new CWD (measured as CWD with fresh phloem) was considerably reduced in older logging areas: in old-growth forests the volume created by small-scale gap dynamics varied 0.7–1.7 m^3 ha^{-1} y^{-1}, whereas on 15-year old sites the volume was 0.03–0.1 m^3 ha^{-1} y^{-1}. Also the proportions of different decay stages changed in the course of time and due to the lack of new CWD: in old-growth stands 30–50% of CWD belonged to mid decay stages, whereas in old regeneration areas their proportion was only 8–10%, and the majority of CWD (60–80%) belonged to advanced decay stage (Sippola et al. 1998).

Among beetles, changes in CWD amount and quality were reflected both in the species composition and species richness. The species richness of saproxylics increased on the recently logged sites compared with old-growth forests, however, the difference was not significant due to the large variation among the sites (Table 1). The increase in the number of saproxylics was due to the emergence of many primary colonizers of CWD, e.g., many cambial feeders and their associates, which were attracted by logging waste. On the 15-year old sites, the mean number of saproxylics was at the same level as before the logging. The number of non-saproxylics also inreased after logging, and was significantly higher in the 15-year old logging areas than in the old-growth forests (Table 1). The increase was due to many species which prefer open habitats and primary successional stages (Sippola et al. 2002).

In the old-growth spruce forests the average volume of CWD was 19.4 m^3 ha^{-1}, but in the 15-year old clear-cuts only 8.3 m^3 ha^{-1}. No differences were observed in the species numbers of beetles between old-growth and clear-cut sites. The species composition, however, was considerably different between natural and managed stands, and they were clearly separated from each other in the DCA-ordination. The most abundant saproxylic species of old-growth forests, such as *Hylecoetus dermestodes, H. flabellicornis*, *Hylurgops glabratus* and *Hylastes cunicularius* were almost totally absent from the clear-cuts. Also many other species of old-growth spruce stands had disappeared or were less abundant in clear-cuts, most of these were species which are dependent on different microfungi. Instead, species which prefer open areas, many elaterids which live in soil as larvae, and some pine-inhabiting species had emerged or increased in abundance (Sippola et al. 2002). The proportion of rare species was considerably lower in the clear-cuts than in the virgin spruce forests (Table 1).

The correlation analysis revealed three groups of environmental variables that correlated positively with the total species richness of beetles in old-growth forests: (1) The site fertility, which was indicated by correlations with the number of vascular plants, cover of eutrophic vegetation, thickness of humus layer and the total volume of living spruce, (2) total volume of CWD, and (3) the quality of certain CWD types (CWD of spruce and deciduous trees, and decay stages 3 and 4). None of the environmental variables correlated positively with the species richness of beetles on the regeneration sites. Two threatened species, *Agathidium pallidum* and *Pytho abieticola* were found in the beetle study, both from old-growth spruce-dominated stands.

3.2 Effects of seed-tree and selective cutting on polypore diversity

The effects of seed-tree cutting on polypore diversity were studied partly in the same areas which were used in the beetle studies, but to get a longer time perspective, areas that had been logged about 40 years earlier were also included in the study. The volume of CWD on the 40-year old sites was only slightly lower than on the 15-year old sites, indicating that decomposition of large-diameter CWD is very slow in the areas near the timberline. The accumulation of new CWD was still lowered on the 40-year old sites, being 0.004 m^3 ha^{-1} y^{-1}.

The main separating factor between the species compositions of the sites in the DCA-ordination was the successional stage of decaying wood, which separated the 3-year old sites from other study sites. The species composition on these sites was dominated by early decomposers, whereas the species composition on older logging sites resembled that of old-growth stands. The expected species number of polypores for a given sample size was

Table 1. CWD volumes and numbers of beetle and polypore species in different forest site types and management areas in boreal forests. The CWD volumes are m^3 ha^{-1}, and the numbers of species means per site, if not indicated otherwise.

	Old-growth forests	1–3 years old seed-tree cut pine forests	15–18 years oldseed-tree cut pine forests	40–43 years old seed-tree cut pine forests	15–years old clear-cut spruce forests	60–100 years old selectively logged spruce forests	WKHs with rich vascular plant flora
Beetle diversity in northern boreal pine forests							
Total CWD	18.8	25.7	17.4				
CWD > 30 cm	2.8	0.7	0.2				
Number of saproxylics	19.2	24.1	19.0				
Number of non-saproxylics	15.9	23.3	22.9				
Rare species (% of all species)	6.7	6.7	2.8				
Beetle diversity in northern boreal spruce forests							
Total CWD	19.4				8.3		
CWD > 30 cm	3.3				1.3		
Number of saproxylics	19.5				19.9		
Number of non-saproxylics	17.8				18.0		
Rare species (% of all species)	7.6				2.5		
Polypore diversity in northern boreal pine forests							
Total CWD	18.8	25.7	17.4	14.8			
Accumulation of new CWD	0.7	9.7	0.03	0.004			
Number of species on logging waste	-	9.0	13.0	7.5			
Number of species on pre-logging/natural CWD	14.3	12.5	11.0	13.5			
Number of indicator and threatened species	3.0	2.0	3.5	3.5			

Table 1. Continued.

	Old-growth forests	1–3 years old seed-tree cut pine forests	15–18 years oldseed-tree cut pine forests	40–43 years old seed-tree cut pine forests	15–years old clear-cut spruce forests	60–100 years old selectively logged spruce forests	WKHs with rich vascular plant flora
Polypore diversity in northern/middle boreal spruce forests							
Total CWD	51.3					29.1	
CWD > 30 cm	18.3					10.0	
Volume of logs	34.9					15.3	
Number of species	29.6					17.2	
Number of indicator and threatened species	9.0					4.0	
Records of indicator and threatened species	53.6					20.0	
WKHs in southern boreal spruce forests							
Total CWD	67.2						40.9
Volume of coniferous CWD	45.3						29.6
Volume of deciduous CWD	21.9						11.3
Number of species per 35 study plots	34.3						29.8
Proportion of species on coniferous CWD (% of all species)	47						23
Proportion of species on deciduous CWD (% of all species)	42						71
Number of indicator and threatened species per study plot	1.4						0.3

significantly higher in the old-growth pine forests than in the 40-year old stands. However, when the species living on pine CWD and birch CWD were considered separately, it was observed that the difference was due to the species living on deciduous CWD, whereas no differences were found in the numbers of species living on pine CWD (Sippola and Renvall 1999). When the species living on pre-logging CWD (which had been created before the logging) and on the logging waste were analysed separately, it was observed that the pre-logging CWD in 40-year old areas hosted almost the same number of species than the old-growth stands, but the number of species on logging waste was only about half of the species number found on pre-logging CWD (Table 1). Of the eight threatened species found in the study, four could also invade logging waste (*Antrodia albobrunnea*, *Postia hibernica*, *P. lateritia* and *Skeletocutis stellae*). Three species were found also in logging areas, but exclusively on pre-logging CWD (*Skeletocutis lenis*, *Antrodia infirma* and *Gelatoporia pannocincta*), and one species was recorded only in old-growth forest (*Skeletocutis jelicii*) (Sippola and Renvall 1999).

The spruce forests that had been selectively logged 60–100 years ago were located at the border of middle and north boreal zones in Kainuu. Logging at that time was directed to the largest trees of the stands, and all the coniferous trees which exceeded certain minimum diameter at breast height were harvested (Karjalainen 1998). The sites were compared with nearby spruce stands that had been completely left out of logging. The results showed that the forest structure and CWD volume of the logged stands had not returned into the natural level in the time span of 60–100 years. The mean volume of living trees was 22% lower than in the primeval forests, this difference was, however, not statistically significant. The volume of CWD, instead, was on average 43% lower on the logged sites (Table 1), and the differences in the mean volumes were statistically significant. The differences in the mean numbers of polypore species per site were statistically significant between the primeval and logged sites (Table 1), and the total number of species correlated negatively with the number of cut stumps. With further analysing, the difference was observed in the number of species inhabiting spruce logs, whereas no differences were found in the numbers of species inhabiting deciduous CWD or snags, stumps and branches. We found altogether seven threatened species, of which four exclusively in primeval forests (*Skeletocutis odora, Postia lateritia, Antrodia albobrunnea* and *Junghuhnia collabens*), one exclusively in selectively logged forest (*Skeletocutis lilacina*) and two in both (*Diplomitoporus crustulinus, Skeletocutis stellae*). In addition, eight species which are regarded as old-growth forest indicators (according to Kotiranta and Niemelä 1996) were recorded. The combined number of threatened and old-growth forest indicator species per site, as well as their number of records, were significantly higher in primeval forests than in the logged stands (Table 1). The species showed different ecological tolerance for logging: species that prefer large-diameter trunks and which have found to be sensitive to environmental changes such as *Fomitopsis rosea* and *Amylocystis lapponica* (Renvall 1995; Bader et al. 1995; Bredesen et al. 1997) were mainly absent from the selectively logged areas, whereas many species, which are able to inhabit relatively small-diameter CWD were found on the logged sites. The number of indicator species showed positive correlation with the volume of snags, and negative correlation with number of cut stumps (Sippola et al. 2001).

3.3 Importance of woodland key habitats for polypore diversity

Woodland key habitats are defined as actual or potential sites for threatened or rare species. They are assumed to be shelter and source areas for species that have suffered from forestry operations (Snäll and Jonsson 2001). Usually WKHs are relatively small in size, for instance in Finland their mean size was 0.21 ha in the state-owned and 0.23 ha in the private forests in 1999 (Hänninen 2001). In our study area the coverage of living spruce was similar between the control and WKH sites, but there was a larger number of small-diameter deciduous trees in the WKHs, especially aspen, rowan and bird cherry. The total amount of CWD was significantly lower in the WKHs than on control sites (Table 1). When equal sample sizes (number of study plots) between controls and WKHs were compared by resampling, the mean number of species in control areas was significantly higher than in the WKHs (Table 1). The species number of WKHs covered about 60% of the species number of control sites. The most significant difference was that about 70% of the species in WKHs were species of deciduous CWD, whereas in control areas the number of species on deciduous and coniferous CWD were about equal. In this study, a 0.2 ha-size WKH covered 19% of the total species richness found in the study, and a 0.5 ha-size WKH about 40% . Only three threatened species were found: *Amyloporia sitchensis* and *Protomerulius caryae* from the control sites and *Antrodia mellita* from a WKH. The number of old-growth forest indicators was significantly higher in the controls than in the WKHs (Sippola et al. 2004).

4. Discussion

Beetle species of old-growth pine forests seemed to be relative tolerant to environmental changes caused by seed-tree cutting. Even though there were changes in the species composition after logging, the majority of forest species seemed to be able to inhabit regeneration sites. This may be related to two factors: first, the old-growth pine forests in the northern latitudes are relatively open and sparse naturally, and the change in microclimate after logging is relatively small. Secondly, pine forests in boreal zone are naturally subjected to relatively frequent disturbances by fire (Zackrisson 1977; Engelmark 1984; 1987), and the forest species probably are adapted to these regular disturbances. The proportion of rare species, however, was much lower on the 15-year old seed-tree cut sites than on the old-growth sites. The obvious reason for this is the reduced volume of CWD in these areas and the subsequent lack of mid decay stages, which have been found to host the highest number of rare and threatened beetles in boreal zone (Esseen et al. 1992; Jonsell et al. 1998).

Clear-cutting, ploughing of soil and planting the areas with pine is a very common forest regeneration method of spruce forests in Finland, partly because pine has been considered more valuable than spruce as timber, and partly in order to prevent root-rot disease and to improve the soil properties (Hannelius and Kuusela 1995). The results of the beetle study show that even though the species richness in the clear-cuts remained as high as in the old-growth spruce stands, the species composition was drastically changed already 15 years after the logging. In this case, both the change in the amount and quality of substrate, and the change in microclimate affected, since the species groups that were mostly affected were those confined to spruce and different microfungi, which evidently suffer from drought in the clear-cuts. The same reasons obviously affected to the low proportion of rare species on these sites.

The diversity of polyporous fungi remained high still 40 years after logging in pine forests. However, the species richness was maintained mainly on pre-logging CWD, whereas the logging waste could host only about half of the total species diversity of old-growth forests. About 36% of the total number species was found on CWD with the mean base diameter > 20 cm. Because logging waste is usually small-diameter, it both decomposes faster and seems also to be unsuitable for many old-growth forest species which prefer large-diameter trunks as their substrate (Bader et al. 1995; Renvall 1995). On the other hand, the result indicates that it is rather the lack of suitable substrate than mircoclimatic change which is the major reason for the low polypore diversity in pine-dominated regeneration areas. The result also shows that there is a time lag of several decades, even a hundred years, before the real effects of logging on saproxylic and saprotrophic species can be detected. It has been estimated that this extinction debt will further strongly decrease species diversity in regeneration areas in boreal forests (Hanski 2000). This is probable especially in northern regions, where tree growth, and, consequently, production of new large-diameter CWD is slow. For instance, the mean diameter (DBH = 1.3 m) of pine in Finnish Lapland in 100 year old stands is 12 cm and in 200 year old stands 18 cm (Gustavssen and Timonen 1999). Continuous supply of new large-diameter CWD would allow a large portion of saproxylic and saprotrophic species to survive in managed pine forests. Besides of leaving large-diameter CWD on the sites during logging, this could be ensured by leaving part of the seed-trees unlogged and even by creating CWD artificially (cf. Lindhe 2004).

Polypore studies in the old selectively logged stands show that logging, which had been directed to the largest trees of the stands had decreased the number of polypore species, and neither the forest structure nor the species composition had returned to natural level in 60–100 years. However, many old-growth forest indicators and some threatened species could survive in the selectively logged areas if suitable substrate was available. The large-scale inventory of old-growth forests in northern Finland 1993-94 showed that old selectively logged forests still maintained relatively high species diversity, including many indicator and threatened species, when compared with other managed forests (Ministry of Environment 1996; Kumpulainen et al. 1997). The probable reason for this is that the canopy density has been largerly maintained in selective logging. Some unlogged spots, which may have served as species sources, have also been left unlogged during the old times, when efficient machines could not be used and the transport of logs was by horse.

Direct comparison of the results of beetle and polypore studies can be questioned, because polypore occurrence is detected directly from the fruit bodies indicating that CWD is inhabited by the species, but beetles were trapped by free-hanging window traps, and the result may be affected by the flight activity of species, random drift of a specimen by air currents etc. Besides, the method is not catching non-flying species. However, compared with other beetle trapping methods window flight trapping yields the largest number of species, and gives a relatively good picture of saproxylic species (Siitonen 1994, Økland 1996, Similä et al. 2002 b). Økland (1996) estimates that the method is better suited for comparisons of beetle assemblages between different forest environments than trunk-window traps, but it collects lower number of rare and threatened species than the latter method (Muona 1999, Martikainen 2000).

The results of both beetle and polypore studies indicate that maintaining the original species diversity of spruce stands in managed forests is difficult. Retention trees and CWD left on the logged sites will help to preserve part of the saproxylic diversity, and, as shown by Lindhe (2004), also artificially created logs and high stumps maintain a large portion of saproxylic species even on clear-cut sites. Our studies indicate, however, that those species which are adapted to moist, stable conditions of old-growth spruce forests will be absent from the managed stands for a long period of time, and, if the tree species is changed or the source areas are distant, may be extinct from the stand (cf. Högberg 1998; Nordén 2000, Edman et al. 2004 a,b). Modern selective logging ("continuous cover forestry"), where naturally uneven stand structure is maintained by removing trees of different age classes might best maintain species diversity of spruce forests. In the areas with intensive forestry, unlogged patches of 0.5-1 ha within the logged areas could serve as source areas for species.

Woodland Key Habitats in Koli National Park had been distinguished on the basis on their rich vascular plant flora. Results of the polypore study showed that the high species richness of vascular plants did not indicate high polypore diversity. This is in harmony with the results of most of the studies on the co-variation of species richness in boreal forests, which show no co-variation with the vascular plant and polypore species richnesses (Jonsson and Jonsell 1999; Virolainen et al. 2000; Similä et al. 2003). Sætersdal et al. (2003), however, found that in small-size plots the species richness of vascular plants correlated with the species richnesses of several groups, including polypores. Due to the differences in forest strucure, the WKHs in the Koli National Park preserved mainly species of deciduous trees, whereas typical species of old-growth spruce forests were poorly maintained. Small WKHs maintained species diversity randomly: patches of 0.2 ha included only about 20% of the detected total species richness of polypores. Both the total number of species and the numbers of threatened and indicator species increased strongly up to one hectare size. Increase of the mean size of WKHs would benefit the survival of polypore species, and including also patches of typical old-growth forests would help to maintain the total species richness of the each region.

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Contributions of New Remote Sensing Tools to Forest Biodiversity Assessment and Monitoring

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Abstract

In this paper, the potential application of relatively new remotely sensed data in the context of biodiversity assessment and monitoring is presented. Although other authors have partially covered the topic, this paper aims to complement available information by presenting relatively new advances in the satellite technology and its applications. A conceptual framework for the application of remotely sensed data in biodiversity monitoring at different spatial scales is proposed. Special attention is paid to the new hyper spatial resolution sensors and laser data.

Keywords: aerial photography; landscape management; laser; sustainability

1. Introduction

Because the current decline in biodiversity has been identified as a major environmental issue, the relevance of assessment and monitoring is apparent throughout the world. This concern has been recognized in international initiatives such as the Convention on Biological Diversity, as well as EU Biodiversity Strategy. The development of indicators that allow a precise monitoring of biodiversity change is one of the current scientific challenges, considering that monitoring needs to occur at a variety of scales in order to supply the information required on specialist species, critical habitats, mortality, recruitment and landscape-scale patterns (Pierce and Ervin 1999).

The most prevalent usage of the term "biodiversity" is as a synonym for the "variety of life" (Gaston 1996). This abstract concept, expressed across a range of hierarchical scales, cannot be encapsulated in a single variable (Gaston 1996). The complexity in this sense is irreducible. The species is regarded in many quarters as the fundamental unit of biodiversity,

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species richness as the fundamental meaning of biodiversity, and the high level of species extinction as the main manifestation of the biodiversity crisis (Gaston 1996). Species richness is at best a measure of one aspect of biodiversity. In that species are by definition different in some sense, species richness must capture some facet of this variety (Heywood 1995).

It has also been argued that landscape structures themselves are components of biodiversity (Heywood 1995). Therefore, and for the purposes of this paper, biodiversity is an encompassing term for the diversity of landscapes, species, populations and genes (Heywood 1995).

Natural resources management requires synoptic, repetitive, quantitative and spatial biophysical and vegetation data over large geographic areas and over long periods. Remote sensing is the only way to acquire such data (Franklin 2001) and it has proven to be an efficient tool in the context of biodiversity studies (Innes and Koch 1998). In this paper, the potential as well as the application of remotely sensed data for assessment and monitoring of biodiversity is presented. Although other authors have partially covered the topic (e.g., Innes and Koch 1998; Koch and Ivits 2002), this paper complements the available material by presenting relatively new advances in the satellite technology and selected applications.

2. Monitoring Biodiversity: A Conceptual Framework

Due to the practical and conceptual difficulties of measuring changes to biodiversity in forests, and the uncertainty of indicators/keystone species (the selection of the keystone species is rather arbitrary and their role needs to be demonstrated (Noss 1999)), changes in biodiversity may be assessed indirectly through the assessment of the process that maintain and generate it, as Stork et al. (1997) suggest. Figure 1 depicts a conceptual model concerning the influence of human activities on biodiversity and the possible role of remote sensing data (Stork et al. 1997).

The direct effects of human activities on forests produce impacts on the processes that generate and maintain biodiversity, including changes in forest area, transformations to non-forest vegetation types or transformations to other vegetation types (Stork et al. 1997). The fragmentation effect implies a change in the spatial mosaic of the forest, which implies a modification in patch connectivity and/or in the length and complexity of patch edges, among others. These spatial changes can affect the ability of an organism to move within a landscape (Bennett 1999). Pollution can change ecological processes such as reproduction, predator-prey relationships and nutrient cycling (Primack 1993). Some human interventions cause a direct loss of species and a change in nutrient conditions, which can affect processes that influence biodiversity (Stork et al. 1997). Moreover, direct effects of human interventions (Figure 1) alter the processes that generate and maintain biodiversity, including the dispersal and migration of species, reproduction, trophic dynamics, ecosystem processes, local extinction, regeneration and succession, as well as natural disturbance regimes (Stork et al. 1997, Figure 1)

Remote sensing can contribute in various ways to the assessment and monitoring of biodiversity by analyzing the different components of the processes that maintain biodiversity, as mentioned above (Figure 1). Remote sensing has proven to be a useful tool for monitoring the direct effects of human interventions such as fragmentation, area change and pollution (Figure 1), as discussed later. In terms of the processes that maintain and shape biodiversity, the role of remotely sensed data is limited to providing information on regeneration, successional processes and natural disturbances (e.g. fire, floods).

Indicators of biodiversity change based on remotely sensed data, such as those presented by Stork et al. (1997) or those mentioned by Pierce and Ervin (1999), can be evaluated using this conceptual model, and as a complement or an alternative to the indicator species (Table 1).

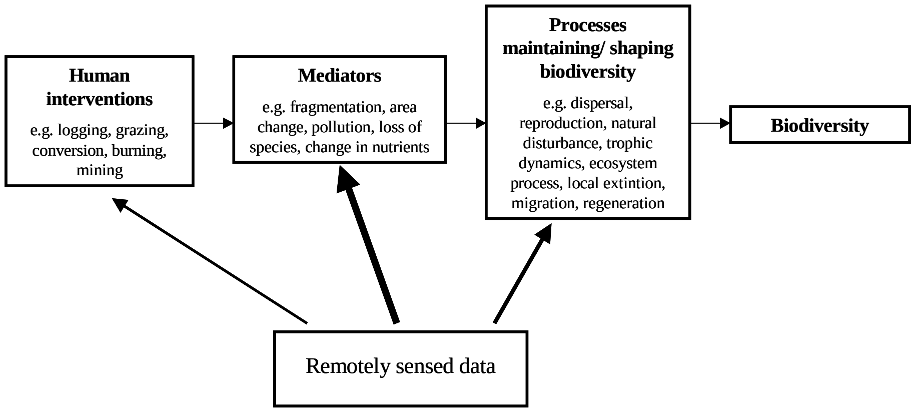

Figure 1. Possible role of remotely sensed data according to a model that defines the relationships between human activities, mediating processes, ecological processes and biodiversity. Note that the thickness of the arrows indicates the importance of the remotely sensed data in monitoring the process. Adapted from Stock et al. (1997).

3. Remotely Sensed Data for Forest Biodiversity Assessment.

Remote sensing can be considered to refer to spaceborne or airborne technologies. The first term refers to digital satellite data, while airborne sensing methods may be either aerial photography or digital data.

Hyperspectral Imagery

These relatively new sensors (also known as Imaging Spectrometers) measure earth materials and produce complete spectral signatures with no wavelength omissions (Franklin 2001). Such instruments are flown aboard space or air-based platforms. This type of data is conceived as the next step in the spectral dimension of the evolution of multispectral imaging radiometers, such as the Landsat data for example (Birk and McCord 1994).

Several airborne systems have been developed and flown over the past twenty years, such as the Compact Airborne Spectrographic Imager (CASI) and the Airborne Visible/Infrared. Imaging Spectrometer (AVIRIS), with 224 bands between 400 and 2500 nm (AVIRIS 2003). Improvements have been made in signal-to-noise ratio, radiometric and spectral calibration accuracy, sensor size, swath width and number of spectral channels.

Microwave Sensors

Radar (Radio Detection and Ranging) imagery can be used as a multipurpose data source for forestry applications, especially in areas of severe cloud limitations (Bush and Ulaby 1978). The most common mode of operation for active microwave sensors is Synthetic Aperture Radar (SAR), which is exclusive to moving platforms. Radar data has been successfully applied to the

Table 1. Proposed indicators of landscape pattern by Stork et al (1997).

Area	Extension	• Areal extent of each vegetation type in the intervention area relative to area of the vegetation type in the total area. A decrease in the total area of habitat available often correlates with species decline.
Area	Extension	• Areal extent of each vegetation type in the intervention area relative to area of the vegetation type in the total area. A decrease in the total area of habitat available often correlates with species decline.
Fragmentation	Patch structure	• Number of patches of each vegetation type per unit area/ concession. The number of patch types present is important because many organisms are associated with a single type, and thus patch richness may correlate with species richness. • Largest patch size of each vegetation type. Information on maximum patch size may provide insight into long-term population viability because populations are unlikely to persist in landscapes where the largest patch is smaller than that species' home range. • Area-weighted patch size. This verifier reflects the average patch size/total area for each vegetation type • Contagion. The contagion index measures the extent to which land covers/vegetation types are clumped or aggregated. Contagion is a useful metric for those species that require large contiguous areas of a particular land cover. • Dominance. It measures evenness, in contrast to richness of patch structure. Its value indicates the degree to which species dependent on a single habitat can pervade the landscape. • Fractal dimension. It provides a measure of complexity of patch shape. Natural areas tend to have a more complex shape and a higher fractal value than human-altered landscapes
	Connectivity	• Average, minimum and maximum distance between two patches of the same cover type. This is a measure of distance between patches. • Percolation index. This measures the connectedness of a landscape from one edge to the other.
	Edge features	• Linear measure of the total amount of edge of each vegetation type. The length of edge between different land-cover types is useful for assessing habitat for species that prefer or avoid certain types of eco-tones, and can change processes such as predation rates. • Amount of edge around the largest patch. To the extent that the largest patch has significance, its perimeter can provide a measure of diversity.

assessment of forest areas (Ramminger et al. 2002), but as recently pointed out by Franklin (2001) no standardized methods with reproducible results are available. However, it is expected that planned sensors (e.g. Radarsat-2) will provide effective SAR data for forestry applications (Franklin 2001). Relatively new sensors, such as the airborne SAR system Carabas, produce very high-resolution (ca. 2.5 m) images and have even demonstrated the potential to detect individual trees in very sparse forests (Carabas Forestry 2003).

Lidar (LIght Detection And Ranging) is similar to the more familiar radar, and can be thought of as laser radar (Dubayah and Drake 2000). In a radar, radio waves are transmitted into the atmosphere, which scatters some of the power back to the radar's receiver. A lidar also transmits and receives electromagnetic radiation, but at a higher frequency. Lidar sensors operate in the ultraviolet, visible and infrared region of the electromagnetic spectrum (Dubayah and Drake 2000). A new scanning lidar sensor, known as SLICER (Scanning Lidar Imager of Canopies by Echo Recovery), has been deployed in studies of forest structure, biomass and canopy volume (Franklin 2001). This sensor differs from the earlier-generation lidars in that the entire laser return signal is digitized (Franklin 2001). The upcoming Vegetation Canopy Lidar (VCL) (scheduled for 2003, VCL 2003) will provide a global data set of canopy height, vertical distributions of intercepted surfaces and subcanopy topography, all of which have remarkable potential for biodiversity applications, such as habitat monitoring.

Multispectral Imagery

Due to the technical difficulties associated with the processing of radar and lidar data, the majority of studies have been concentrated on applications that utilize optical technologies (Innes and Koch 1998). However, the well-known limitations associated with the use of satellite data in terms of scale (pixel size) and stereoscopy information have decreased in the recent years. The new hyperspatial resolution images, with better geometric resolution and temporal frequency, have increased the potential of remote sensing in investigations of forest biodiversity at regional, landscape and local scales; as discussed in the next section. New optical hyperspatial resolution images, such as IKONOS (5 bands and 4m spatial resolution multispectral, 1m resolution panchromatic channel) and Quickbird (5 channels, 0.61 m spatial resolution panchromatic band, 2.5 m spatial resolution multispectral), provide spatial resolutions that can be measured in centimeters as well as stereoscopy capability (e.g., SPOT-5, 4 channels 10 m spatial resolution multispectral, 5 m and 2.5 m panchromatic band), which opens a new range of possible applications (Franklin 2001).

4. New Contributions to Biodiversity at Landscape and Stand Scales

Forests are embedded within a landscape consisting of a variety of other habitats, leading to a mosaic of different habitats (Forman 1995). The spatial forest structure can be quantified from observations of the composition and structure of landscape patches, which are directly related to ecosystem and population processes (Forman 1995) that have been mapped from elements obtained by remote sensing (Franklin 2001). Forman (1995) describes five disturbance processes that change the landscape, influence habitat loss and that can occurs simultaneously:

- Perforation: the creation of holes in the patch or landscape.
- Dissection: cutting a landscape area or matrix into equally wide linear features, such as roads.
- Fragmentation: breaking and separating the matrix into smaller, non-contiguous segments or patches.
- Shrinkage: the sizes of patches decrease.
- Attrition: patch disappearance.

Remotely sensed data enables the measurement and tracking of features directly related to the above-mentioned disturbance processes. A large quantity of indices has been developed for such purposes, but according to Baskent and Jordan (1995) three general groups of indices can be identified:

- *Areal indices*. This set of indices provides a measure of landscape or patch size, shape and interior or core area.
- *Linear indices*. These indices provide a measure of boundary length, width and shape at the patch level, as well as connectivity and circuitry at the landscape level.
- *Topological indices*. This set of indices provides measures of the spatial relationships between landscape elements in terms of dispersion, spatial association, interspersion, isolation and connectivity.

A large number of studies examine the utility of these landscape indices for biodiversity assessment and monitoring. Software used to calculate landscape metrics has been available for some time and software modules for landscape analysis have recently been integrated into commercial GIS software. Problems associated with the factors affecting the indices selection (pixel size, area, multi-collinearity, among others) have been discussed by Innes and Koch (1998).

In the case of hyperspectral sensors, Stone et al. (2001) demonstrate the potential of these types of sensors for the development of robust forest health indicators, whereas Buckingham et al. (2002) utilized this type of data to discriminate forest ecosystems and species identification, productivity, leaf water content and canopy chemistry.

This era of new sensor development has led to the rise of a number of new technical problems that were not contemplated before (Schiewe et al. 2001). Though the problem of mixed pixels has been reduced, the internal variability and noise within land-use or land-cover classes due to the high spatial resolution of the images has increased (Schiewe et al. 2001). This situation, along with the characteristic texture exhibited by the images, has made it necessary to develop methods for the extraction of image objects, which are omitted in the pixel-based classifications (Schiewe et al. 2001). Image objects contain (apart from spectral information) attributes such as shape, texture, relational and contextual information that can be used for classification purposes (e.g., Herrera et al. in press). Thus, the combination of hyper resolution images, as well as a precise delineation of landscape components (patches) by means of segmentation techniques (e.g., Herrera et al. in press) will allow the precise extraction of characteristics such as tree species, forest canopy measurements and successional stages that a few years ago could only be extracted accurately from large-scale aerial photos. At the same time, such detail will provide a much better estimation of landscape indices, especially those related to linear metrics, as depicted in Figure 2. Although it has been argued that remote sensing is particularly useful for producing assessments of ecosystem diversity made at landscape scale (Innes and Koch 1998; Noss 1999), new sensors that provide images with a spatial resolution measured in centimeters can allow remote sensing to be utilized in biodiversity assessment and monitoring at local scales (e.g. forest management unit).

The results of Baker et al. (1994) and Luckman et al. (1997) suggest that radar backscatter models, the longer wavelength (L-band) of SAR imagery, may be used to discriminate between different levels of forest biomass up to a certain threshold, and that cross polarized backscatter is more sensitive to changes in biomass density. Radar data also offers the possibility to monitor events that occur in forests, such as large-scale storms, which can also affect the processes that maintain biodiversity (Ramminger et al. 2002).

Lidar instruments have demonstrated the capability to accurately estimate forest structural characteristics such as canopy heights, stand volume, basal area and above ground biomass (Dubayah and Drake 2000) - parameters that can be assessed both at landscape and stand scales. Figure 3 depicts a vertical cut of a forest generated using laser data from the airborne TopySys (Friedlaender 2002).

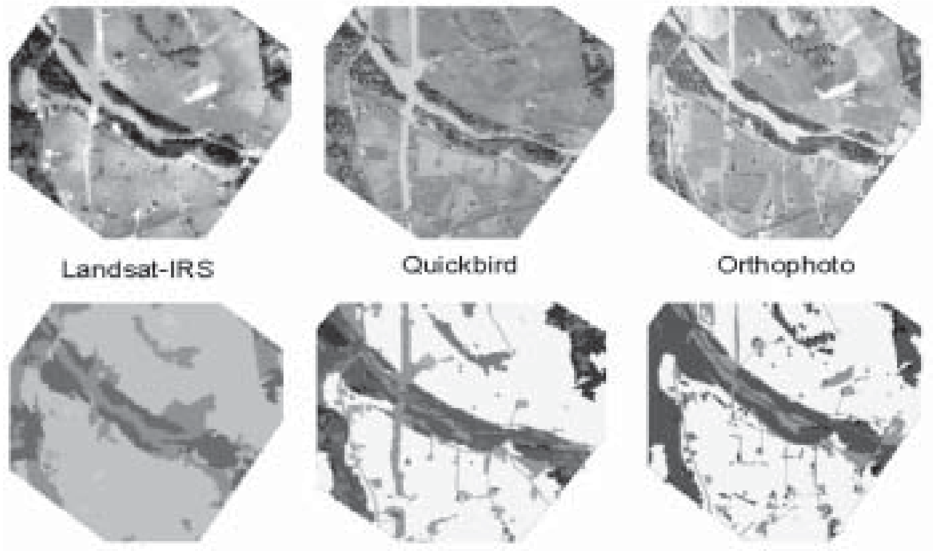

Figure 2. Classification of digital images with different spatial resolutions (from left to right 5m, 2.4m, and 0.60m). Note the difference between 5 and 0.60cm images in terms of the number and shape of the objects extracted (Source: Ivits and Koch in press).

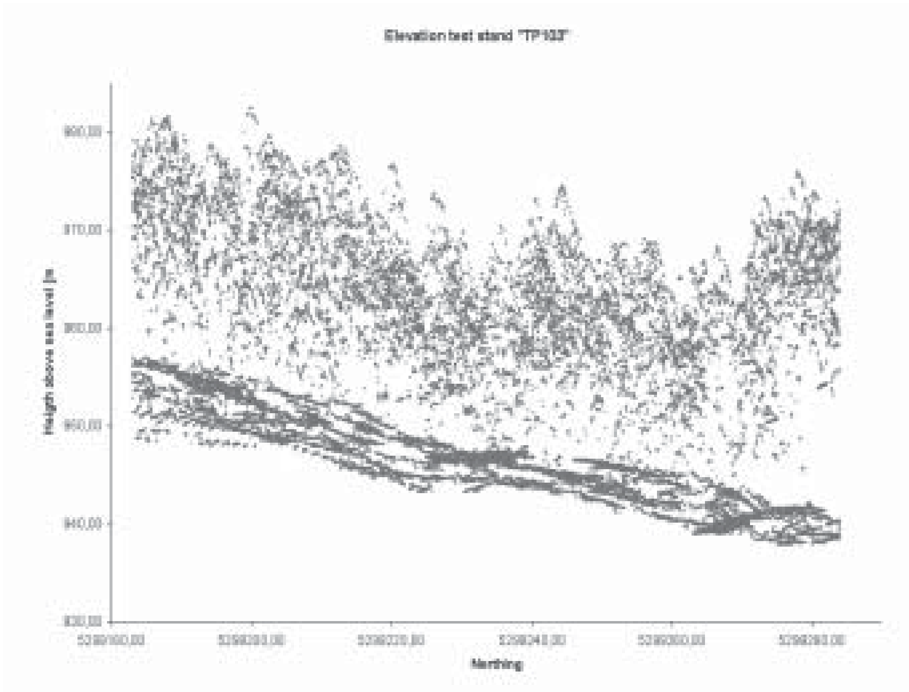

Figure 3. Lasers-scanner data showing a vertical cut through forest. Source: Friedlaender (2002).

Table 2. Potential of lidar remote sensing for forestry applications. (From Dubayah and Drake 2000). D=direct retrieval, M=modeled, I=inferred, F= fusion with other sensors.

Forest characteristic	Lidar derivation
Canopy height	D
Subcanopy topography	D
Vertical distribution of intercepted surfaces	D
Above ground biomass	D
Basal area	M
Canopy volumes	M
Large tree density	I
Canopy cover	F
Life form diversity	F

Harding et al. (2001) studied a successional sequence of four, closed canopy; deciduous forest stands and demonstrated that the SLICER observations reliably provide a measure of canopy structure that reveals ecologically interesting structural variations. Table 2 summarizes the potential application of lidar remote sensing for forestry applications. In forest modeling, this information can be for forest growth estimation (tree-wise growth models), volume estimation, damage estimation, determination of forest evapotranspiration, and forest biodiversity studies.

Regarding the application of multispectral images, new applications relevant to forest biodiversity have been reported. In a recent study carried out in tropical conditions, Bawa et al. (2002) found a positive correlation between species richness and the Normalized Difference Vegetation Index (NDVI), which is an index of green biomass, derived from an IRS-1C LISS III images with a spatial resolution of 23.5 m. The new and planned satellites can play an important role at a local scale, such as the forest management unit. While satellite and airborne data mainly provide information about the canopy surface (Innes and Koch 1998), the new optical sensors such as IKONOS and Quickbird open the possibility to extract information from within the stand. The high geometric resolution, as well as stereoscopy, provides information on major tree species and formation phases. Measurements of crown dimensions and tree height, which were only extractable from aerial photography until a few years ago (Innes and Koch 1998), are now possible.

The use of aerial photography and photogrammetric methods for biodiversity assessments is well recognized in the literature (Innes and Koch 1998). A new classification key allows the extraction of new horizontal and vertical forest parameters from color infrared aerial photos, including the characterization of successional stages, forest edges, gaps and dead wood within forest gaps, among others (AFL in prep.). These parameters are of the most relevance to biodiversity studies, both at local and landscape scales. It is believed that with the combination of hyperspatial resolution images and new multi-resolution segmentation methods commercially available (eCognition), the extraction of these forest structural parameters will soon be available for biodiversity assessment and monitoring at scales of 1:20,000 (AFL in prep.).

5. Final Comments

The monitoring of biodiversity changes at global, regional, landscape and local scales is one of the major tasks of the scientific community. During the last 20 years, remotely sensed data (spaceborne and aerial) has contributed greatly to forest management and biodiversity studies. New hyperspatial images open the development of applications in this task with higher spatial precision. Available and planned radar and lidar sensors will also allow the extraction of relevant features in the context of biological studies. This would not be possible without the parallel improvement of image classification techniques, such as segmentation and fuzzy classification. If these parameters can be extracted from available hyperspatial resolution satellite images – and considering the greater spectral range and sensitivity, near real-time capability and the greater view angle compared to aerial photography – the use of these images will allow cost-effective large-area mapping and underpin the use of remote sensing in ecological studies. However, in the case of hyperspatial resolution images and lidar data, applications at large-scales are still constrained by prohibitive costs and the lack of standardized methods.

Acknowledgements

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Additional Topics

Shrub and Tree Potential as Animal Food in Galicia, NW Spain

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Abstract

The experiment was carried out in Foz, in the northern coast of Lugo (Galicia, NW Spain) and consisted of taking monthly, during a year, samples of fractions with diameter lower than 0.5 mm of shrubs belong to *Araliaceae* (*Hedera helix* L.), Ericaceae (*Calluna vulgaris* (L) Hull, *Daboecia cantabrica* (Hudson) C. Koch and *Erica cinerea* L.), *Leguminosae* (*Cytisus scoparius* (L.) Link and *Ulex minor* Roth), *Rosaceae* (*Rubus ulmifolius* Schott) families, and of tree species from *Betulaceae* (*Alnus glutinosa* (L) Gaertner and *Betula alba* L.), *Fagaceae* (*Quercus robur* L.), *Pinaceae* (*Pinus pinaster* Aiton) and *Salicaeae* (*Populus alba* L.) families, in order to evaluate their potential forage quality (Protein, phosphorus, calcium, potassium and magnesium content). Results showed that *Leguminosae* family has higher protein, phosphorus and potassium content than species from another families. *Hedera helix* and *Rubus ulmifolius* have higher levels of calcium and magnesium than the other shrub species. Populus has the highest level of Ca, Mg and K between studied trees, but protein percentage is higher in *Betula alba* and *Alnus glutinosa* than in the other tree species, that can be explained for *Alnus* by the symbiosis with *Frankia* that can fix nitrogen. *Pinus pinaster* has the highest Na concentration that is low for animal feeding. Taking into account that the diet of animals should not be monospecific, the found results indicate that the levels of those elements will be enough for horse feeding (with the exception of potassium during the winter and the summer) and for goat feeding (with the exception of calcium and magnesium during the winter, and potassium during the winter and spring). Sodium concentrations were usually lower than those described for horse and goat feeding.

Keywords: Shrub quality; forage trees; silvopastoral systems.

Introduction

Galicia has around three million hectares, being the 69.7% occupied by trees plus shrub formations (heathlands). On the other hand, the fifty five percent of regional agrarian rent comes from livestock production. In the last decade an important afforestation process has happened (over 10000 ha per year) and lots of farms have been abandoned; for this reason, in the near future an important forest management effort and economic investments should be made for maintaining forest with optimum tree growth and avoiding natural hazards, like fires (Rigueiro 2000). Some areas of Galicia have a prolonged summer drought (between one and fourth months), when woodland understory and heathlands had an important vegetal development that is transformed in fuel during the summer. For this reason, important and expensive investments are annually made to extinguish fires.

An ecological and cheap alternative that should be take in account for this forest area could be the use of this "fuel" as food for animals. The objective will be that animals will control the vegetal fuel at the same time that meat is produced, increasing the forestland value, as well as the multiproduct option and the sustainable management is developed for improving farm rent and causing rural population stabilisation, all them, ecological, biological, social and economic important aspects promoted by the EU.

There are autochthonous breeds of horses that can be use forest vegetal fuel as food like the "Cabalo galego de Monte breed" that is use to eat this kind of food reducing fire risk and at the same time it is a way of preserving this risk. On the other hand, goat authochthonous breed should be enhanced and promoted in order to increase farm rent from shrubs growing up under forest stand. The use of these "rustic" animals is promoted by Government through subsidies. Knowledge about tree and shrub qualities will help us to know the aptitude as food of the different plants for the different animals.

This study should be made in different parts of the region of Galicia as it is in a transitional area between Atlantic and Mediterranean areas. On the other hand tree leaves intake can be an important source of nutrient in the food shortage period (summer) as a forage-tree, at the time that helps natural pruning.

The objective of this study was to evaluate the concentration of the main macroelements in some shrubs and trees common in Galicia mountains.

Material and Methods

The experiment was carried out in Foz (43º 32´ 26´´ N, 07º 04´ 40'' S, NW Spain) at a 43 m of altitude. Mean annual temperature is around 12–14 ºC, the total annual precipitation per year is between 1200 and 1400 mm, and the summer drought period is around two months.

During 1999 and 2000, samples of ten tree leaves taken from the lower branches of the trees and leaves and woody twig fraction (less than 0.5 cm diameter) of ten shrubs were randomised taken monthly, by using pruning shears, and phenology was evaluated, meaning flowering from the flowering bud appearance to flower disappearance. Shrub branches cut at 0.5 cm diameter do allow better animal growth. Site soil characteristics (pH and organic matter) can be seen in Table 1.

Soils were mainly acid or very acid, with the exception of those were *Alnus glutinosa*, *Populus alba* and *Quercus robur* were developed, that were neutre. Samples were transported to laboratory, dried (48 hours at 40 ºC) and mowed. Nitrogen, phosphorus, potassium, calcium and magnesium were estimated by spectrophotometry after micro-kjeldahl digestion (Castro et al. 1990). ANOVA was used for statistical analyses, with time replicates, and means were separated by Duncan test.

Table 1. pH (water 1:2.5), organic matter (%) and site description, where the trees and shrubs species were sampled.

PH	OM (%)	Description	Shrubs	Tree
6.89	4.70	River border		*Alnus glutinosa*
3.92	18.15	Under *Pinus pinaster* young plantation.	*Daboecia cantabrica* *Erica cinerea* *Calluna vulgaris*.	*Pinus pinaster*
7.15	6.70	Plantation		*Populus alba*
6.80	6.09	Tree border		*Quercus robur*
4.96	8.93	Mixed stand		*Betula alba*
6.19	4.88	Shrubland	*Rubus ulmifolius* *Ulex minor* *Cytisus scoparius* *Hedera helix*	

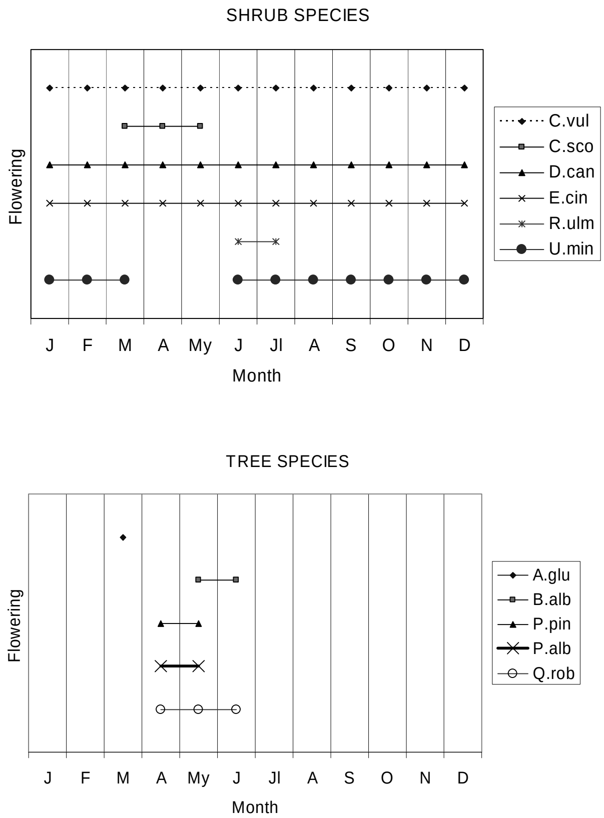

Figure 1. Flowering period for shrub species (C.vul.:*Calluna vulgaris*; C.sco.:*Cytisus scoparius*; D.can:*Daboecia cantabrica*; E.cin.:*Erica cinerea*; H.hel.:*Hedera helix*; R.ulm.:*Rubus ulmifolius*; U.min.:*Ulex minor*; A.glu.:*Alnus glutinosa*; B.alb:*Betula alba*; P.pin: *Pinus pinaster*; P.alb.:*Populus alba* and Q.rob.: *Quercus robur*).

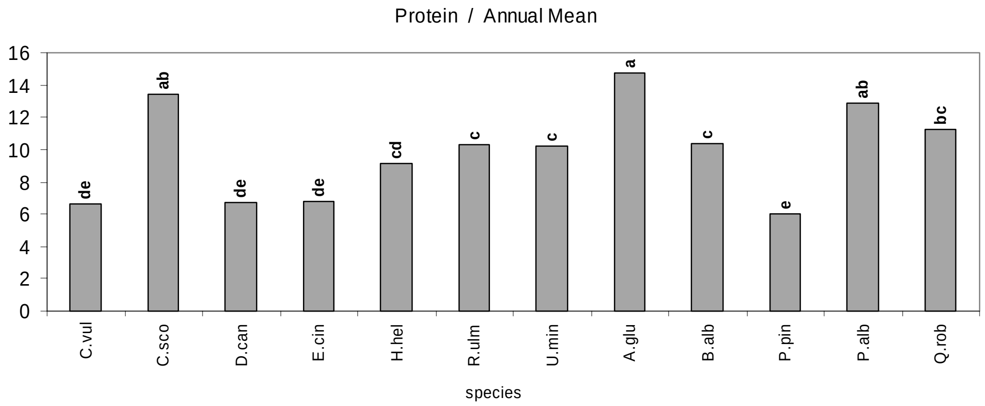

Figure 2. Protein (annual mean) of shrubs (C.vul.:*Calluna vulgaris*; C.sco.:*Cytisus scoparius*; D.can*:Daboecia cantabrica*; E.cin.:*Erica cinerea*; H.hel.:*Hedera helix*; R.ulm.:*Rubus ulmifolius*; U.min.:*Ulex minor*) and tree species (*A.glu.:Alnus glutinosa*; B.alb:*Betula alba*; P.pin: *Pinus pinaster*; P.alb.:*Populus alba* and Q.rob.: *Quercus robur*).

Results and Discussion

Results related to phenology of the studied shrub and tree species can be seen in Figure 1. Tree species had a shorter flowering period than shrub species, which can have a prolonged flowering period or very short (*Rubus ulmifolius*). This prolonged period can be explained because of moderate temperatures due to sea proximity and low altitude, that made that Erica family had some flowered plants during all the year.

Protein percentage of shrub and tree leaves can be seen in Figure 2. Protein annual mean values varied between 6.6 and 13.44% , in shrubs, and 5.75 and 14%, in trees, respectively. These ranges were wider than those found for shrub and tree species described by Rigueiro et al. (2002) in more acid soils. *Daboecia*, *Erica* and *Calluna* had lower protein concentration than other shrub and tree species, with the exception of *Pinus pinaster*, it could be explained because usually flowering period is related to low protein content. Studies developed in the region indicated that values of *Ulex* europaeus and *Cytisus scoparius* were similar that those found here and higher than in the other no-legume species (Gatica et al. 1997; Silva-Pando et al. 1999). Generally, legumes species, as well as *Alnus glutinosa*, had a higher protein concentration thanks to the symbiosis established in their root with micro-organisms that can fix nitrogen. The low protein levels in other species can be explained by the low pH found in soils, which had also a higher OM, as mineralisation is slow and, therefore, nitrogen availability is low. However, shrub protein values found were higher than those described in studies developed in soils with lower pH, with the exception of *Daboecia cantabrica*, and similar to those studies with species grown under trees (Silva-Pando et al. 1999).

Phosphorus mean percentage is showed in Figure 3. Values varied between 0.25 and 0.57%. Shrub phosphorus contents were always higher than those found in more acid soils, described by González et al (1999). However, they were under the range (0.32-2.02) defined by Salcedo et al. (1998) in the spanish mediterranean area, where soils are basic. Tree phosphorus values were always under range described by different authors in acid soils, that could be explained because they took leaf samples in upper branches, that are younger than lower branches (Silva-Pando et al. 2000; Simon and Wild 1998 and Rivero et al. 1999).

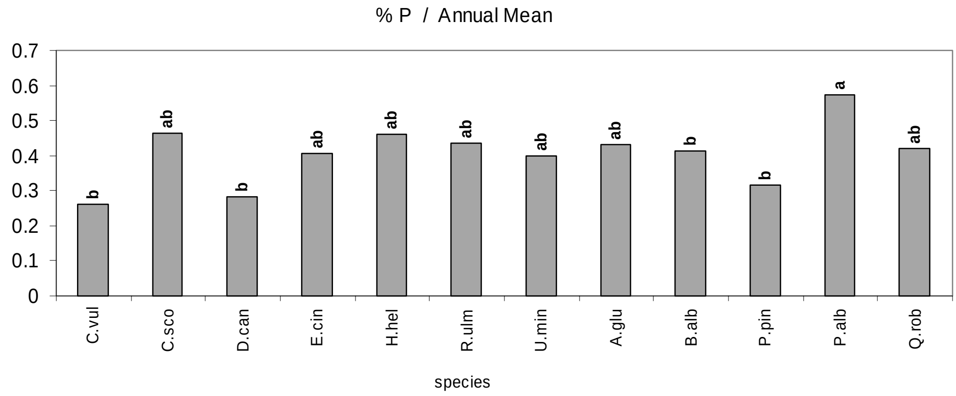

Figure 3. Phosphorus (annual mean) of shrubs (C.vul.:Calluna *vulgaris*; C.sco.:*Cytisus scoparius*; D.can*:Daboecia cantabrica*; E.cin.:*Erica cinerea*; H.hel.:*Hedera helix*; R.ulm.:*Rubus ulmifolius*; U.min.:*Ulex minor*) and tree species (A.glu*.:Alnus glutinosa*; B.alb:*Betula alba*; P.pin: *Pinus pinaster*; P.alb.:*Populus alba* and Q.rob.: *Quercus robur*).

Phosphorus levels showed an important parallelism with protein concentrations, having *Cytisus scoparius* and *Populus alba* the best contents of this element. Phosphorus availability is highly related to pH, being the lowest availability at a low pH, due to the compounds formed with aluminium and iron. On the other hand, lowest phosphorus values ar associated to flowering period.

Horses and goat phosphorus needs for maintaining are 0.2 (NRC 1978) and 0.25% (Arbiza-Aguirre and Oscasberro 1978), that were under the plant values described in this study.

Calcium concentration for shrub and tree species can be seen in Figure 4. Shrub calcium range was between 0.28% and 1.84%, and for trees between 0.31 and 1.40%. Gatica-Trabanini et al. (1997) obtained calcium values for *Cytisus* and *Erica* (2–2.6%) higher than in our case. *Hedera helix* and *Rubus* had higher calcium percentage than Ulex sp., as found González et al. (2000). Simon and Wild (1998) found similar values for Quercus leaves. Lavilla et al. (1999) and González-Hernández (2000) found lower values for *Betula alba*.

Horse and goat needs of calcium for maintaining are 0.3 (NRC 1978) and 0.34 (Arbiza Aguirre 1984), respectively, both limits are covered for most of species, with the exception of *Erica*, *Ulex* and *Pinus pinaster* leaves.

Magnesium mean annual values for shrub and tree species are shown in Figure 5. Shrub concentrations varied between 0.11 and 0.28, in between of that found by González-Hernández et al. 2000 (0.04 – 0.14%) and Gatica-Trabanini et al. 1997 (0.45–0.67%). Tree magnesium percentage interval was close to that referred by González-Hernández et al. (2000). But, Rivero et al. (1999) and Lavilla et al (1999) described magnesium values higher than in our study for *Populus* and *Betula*, respectively. But González- Hernández et al.(2000) indicated lower values than in our case. *Pinus pinaster* concentration was similar to those found by Alonso et al. (1998). Figure 5. Magnesium (annual mean) of shrubs (C.vul.:*Calluna vulgaris*; C.sco.:*Cytisus scoparius*; D.can*:Daboecia cantabrica*; E.cin.:*Erica cinerea*; H.hel.:*Hedera helix*; R.ulm.:*Rubus ulmifolius*; U.min.:*Ulex minor*) and tree species (A.glu.:*Alnus glutinosa*; B.alb:*Betula alba*; P.pin: *Pinus pinaster*; P.alb.:*Populus alba* and Q.rob.: *Quercus robur*).

Horse and goat magnesium needs are around 0.09 and 0.16–0.2, respectively, but, during the winter, *Alnus*, *Pinus*, *Calluna*, *Erica* and *Ulex* did not reach goat need values.

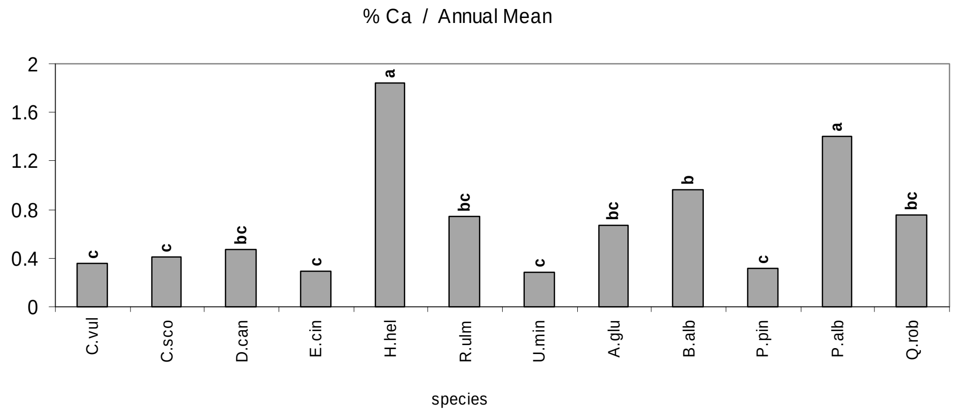

Figure 4. Calcium (annual mean) of shrubs (C.vul.:*Calluna vulgaris*; C.sco.:*Cytisus scoparius*; D.can:*Daboecia cantabrica*; E.cin.:*Erica cinerea*; H.hel.:*Hedera helix*; R.ulm.:*Rubus ulmifolius*; U.min.:*Ulex minor*) and tree species (A.glu.:*Alnus glutinosa*; B.alb:*Betula alba*; P.pin: *Pinus pinaster*; P.alb.:*Populus alba* and Q.rob.: *Quercus robur*).

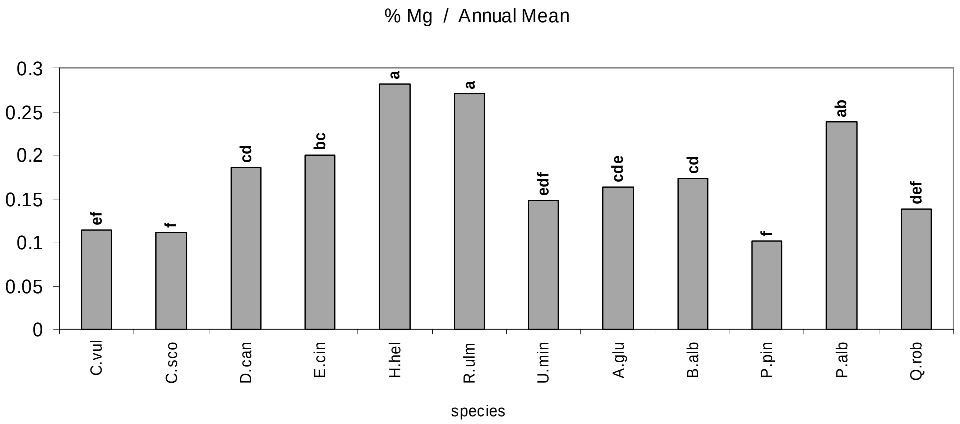

Figure 5. Magnesium (annual mean) of shrubs (C.vul.:*Calluna vulgaris*; C.sco.:*Cytisus scoparius*; D.can:*Daboecia cantabrica*; E.cin.:*Erica cinerea*; H.hel.:*Hedera helix*; R.ulm.:*Rubus ulmifolius*; U.min.:*Ulex minor*) and tree species (A.glu.:*Alnus glutinosa*; B.alb:*Betula alba*; P.pin: *Pinus pinaster*; P.alb.:*Populus alba* and Q.rob.: *Quercus robur*).

Potassium percentages in shrubs (Figure 6) varied between 0.27 and 0.78%, similar values to that found by Gatica et al. (1997) and over that those cited by González et al (2000) (0.17–0.32). Again the genus *Erica* and *Calluna* had the worst values. *Ericaceae* genus are generally associated to acid soils, unfertile and with lower potassium levels (Zas and Alonso 2002). Tree potassium levels varied between 0.57 and 0.67. *Quercus robur* had lower that those cited by Simon and Wild (1998) (0.83–1.11).

Horse maintenance needs for potassium are 0.4% (NRC 1978), whereas, this value is 0.5% (Arbiza-Aguirre 1986) for the maintenance of goats. Most of shrub species did not reach this values; deficiencies are more important during the summer and winter, with the exception of *Hedera helix*, that is eaten intensively by goats.

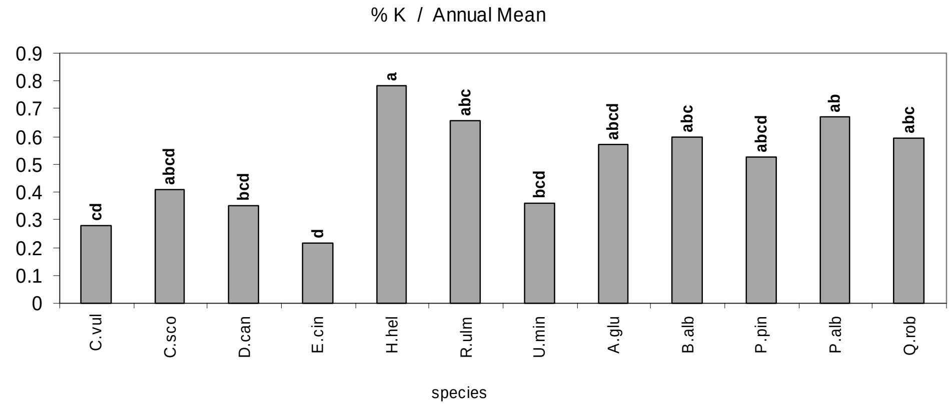

Figure 6. Annual potassium mean of shrub (C.vul.:*Calluna vulgaris*; C.sco.:*Cytisus scoparius*; D.can*:Daboecia cantabrica*; E.cin.:*Erica cinerea*; H.hel.:*Hedera helix*; R.ulm.:*Rubus ulmifolius*; U.min.:*Ulex minor*) and tree species (A.glu.:*Alnus glutinosa*; B.alb:*Betula alba*; P.pin: *Pinus pinaster*; P.alb.:*Populus alba* and Q.rob.: *Quercus robur*).

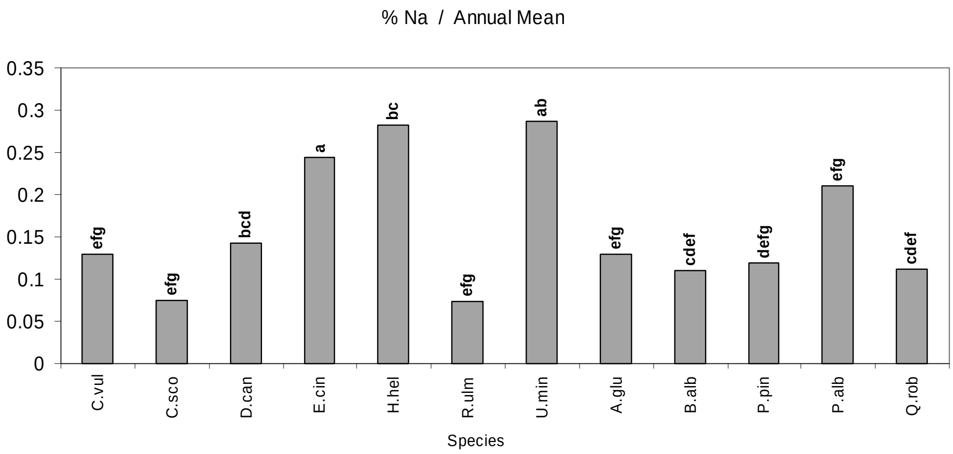

Figure 7. Annual sodium mean of shrub (C.vul.:*Calluna vulgaris*; C.sco.:*Cytisus scoparius*; D.can*:Daboecia cantabrica*; E.cin.:*Erica cinerea*; H.hel.:*Hedera helix*; R.ulm.:*Rubus ulmifolius*; U.min.:*Ulex minor*) and tree species (A.glu.:*Alnus glutinosa*; B.alb:*Betula alba*; P.pin: *Pinus pinaster*; P.alb.:*Populus alba* and Q.rob.: *Quercus robur*).

Sodium shrub and tree percentage (Figure 7) varied between 0.07 –0.28 and 0.11-0.21, respectively. *Cytisus scoparius*, *Calluna vulgaris* and *Rubus ulmifolius* had low sodium content, as well as most of the tree leaves. Horse (0.35%) and goat (0.20%) needs of sodium are not covered by the studied species that makes necessary the supplement with this element.

Relationship K/(Ca+Mg) expressed in miliequivalents can be seen in Figure 8. Hypomagnesemia risk will appear if K/(Ca+Mg) relationship is over 2.2 (Grumes and Allaway 1985). All the shrub species were under this value, but was clearly exceeded by *P. pinaster* and *Q. robur*.

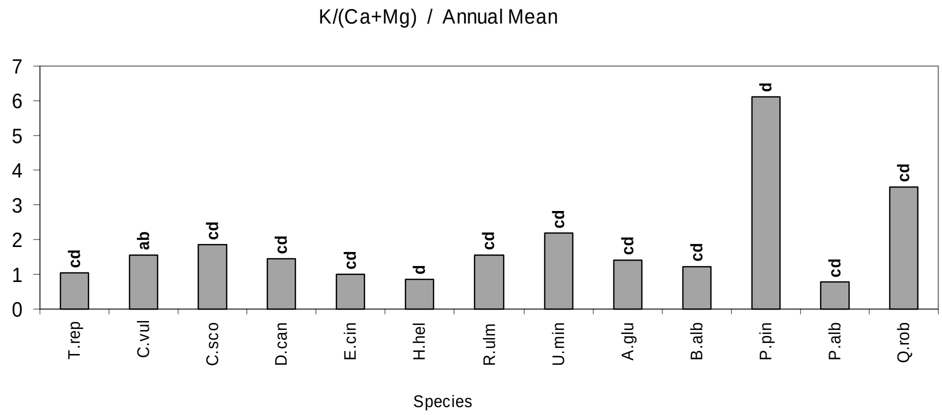

Figure 8. K/(Ca+ Mg) relationship of shrubs (C.vul.:*Calluna vulgaris*; C.sco.:*Cytisus scoparius*; D.can:*Daboecia cantabrica*; E.cin.:*Erica cinerea*; H.hel.:*Hedera helix*; R.ulm.:*Rubus ulmifolius*; U.min.:*Ulex minor*) and tree species (A.glu.:*Alnus glutinosa*; B.alb:*Betula alba*; P.pin: *Pinus pinaster*; P.alb.:*Populus alba* and Q.rob.: *Quercus robur*).

Shrub main nutrients (N, P, K and Ca) were always higher in the *Leguminoseae* than in the ericaceae family. On the other hand, *Ericaceae* species had higher Ca and Mg concentration, being sodium levels similar between both groups, with the exception of *Cytisus*, that had lower values.

Populus had the highest levels of most of the studied elements, with the exception of protein, that was higher in *Alnus glutinosa* leaves. The worst values were found on *Pinus pinaster*, with the exception of sodium that was lower for Betula. This two species are pioneers and adapted to difficult field conditions, like low nutrient levels in the soils.

Taking into account that animal nutrient needs are not precise, because there are not values for rustic animals, like Galician goats and horses, that usually grows up in Galician forestland and that the diet should not be mono-specific, horse needs are covered for all the analysed species, with the exception of potassium during the summer and winter and sodium during all the year. Goat needs will not be covered for potassium during the spring and winter, for calcium and magnesium during the winter and for sodium during all the year.

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The Effect of Fertilisation and Tree Density on *Pinus radiata* Growth and Pasture Production in Silvopastoral Systems in Galicia, NW Spain

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Abstract

The objective of this study was to evaluate the effect of two *Pinus radiata* densities, three types of fertilisation and two sowing mixtures after the first six years of a silvopastoral system established with *Pinus radiata*. Trees were higher under non-fertilised and milk sewage sludge treatments, but differences between treatments for pasture production were found at only a low density. This could be explained by the interception of light at a high density, that limited the pasture production response. The soil pH (water) was higher in mineral treatment at both densities. Differences between treatments were higher at a higher density.

Keywords: Silvopastoral systems; Pinus radiata D. Don; fertilisation

1. Introduction

Galicia is a forest region, approximately 50% of its surface area is occupied by trees, mainly the *Pinus* genus (*Pinus pinaster*, *Pinus radiata* and *Pinus sylvestris*) and *Eucalyptus globulus*. Forest management costs have increased by obligatory shrub clearing for fire prevention as well as fire fighting. These are extremely important in Spain, as 40% of the wooded area burnt in Europe over the last two decades was in Spain (EU 2001). Fire risk is directly related to shrub development under trees (Rigueiro et al. 1999) and shrub encroachment is enhanced by the reduction of forest stand density that favours light input increment into the forest ground. Last silvicultural tendencies promoted the reduction of forest stand density in order to obtain the best wood products, and to make the different silviculture practices with machines easier. Therefore, woodland management aimed at

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increasing herbaceous cover and reducing shrub growth and development under the trees will reduce costs, as clearing is not needed and the risk of fire is avoided. Moreover, if animals can feed on this herbaceous stratum then we will increase income from woodland.

However, management of silvopastoral systems is difficult as light conditions change continuously from when the trees are originally planted to they reaches the mature stage. These changes depend on tree density, fertilisation and how herbaceous species adapt to shade. The objective of this study was to evaluate the effect of two *Pinus radiata* densities, three types of fertilisation and two sowing mixtures after the first six years of a silvopastoral system established with *Pinus radiata*.

2. Material and Methods

The experiment was carried out in Galicia (NW Spain). Twelve treatments were established in the spring of 1995 following a randomised block design with three replicas and they consisted of two densities (833 and 2500 trees/ha), three types of fertilisation (no fertilisation, milk sewage sludge fertilisation (154 m^3/ha, meaning 160 kg total-N ha^{-1}), mineral fertilisation (every year: 500 kg/ha 8:24:16 compound + 40 kg N/ha after second cut) and two types of sowing grass (*Dactylis glomerata* and *Lolium perenne*) (Rigueiro et al. 2000). Milk sewage sludge fertilisation was only applied on establishment and in this treatment, after the fourth year, mineral fertilisation was applied every year. Every experimental unit consisted of 25 trees distributed in a square of 5 x 5 trees, which meant an area of 64 and 192 m^2 for low and high density, respectively. Trees were planted in winter 1999 and yearly measured (height and diameter). Tree pruning took place in the autumn of 2000.

Pasture production (estimated by harvesting the whole area limited by four inner trees), tree growth (estimated by measuring the diameter (caliper) and the height of the 9 inner trees in each plot), botanical composition (100 grams of sward separated by hand), soil pH (water 1:2.5 and ClK) at a depth of 25 cm were determined. ANOVA was used for statistical analyses, and means were separated by the Duncan test method.

3. Results and Discussion

Annual pasture production in each treatment can be seen in Figure 1. Pasture production was higher at a low density and it was significantly affected by density * fertilisation interaction as fertilisation increased pasture production at a low density, but no effect was found at high density.

This is explained by the development of tree canopy (Figure 2) that covers the plots quickly where trees were planted at a rate of 2500 trees/ha, as fertilisation positively affected pasture production during the first years (Rigueiro et al. 2000) at a high density, when no differences were found with the low density (Rigueiro et al. 2000). Therefore, the response of pasture fertilisation depends on density.

Tree height was significantly affected by density, and it was lower in trees which grew with more space, possibly explained by the fact that competition for light between trees at a high density makes trees taller (Figure 2).

Mineral fertilisation treatment negatively affected tree height, probably because it favoured more sward than tree growth in the short term.. Tree diameter was significantly affected by the sown mixture depending on density; at a high density the sown mixture had no effect, but tree diameter was higher when the initial sown mixture was ryegrass at a low density (Figure 2).

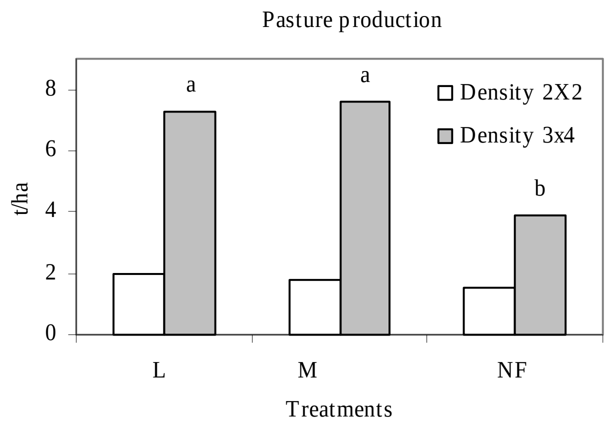


Figure 1. Pasture production in each density (2x2:2500 trees/ha and 3x4:833 trees/ha) and fertilization treatment (L:milk sewage sludge; M:mineral fertilization and NF:No fertilization). Means within density followed by different letter are significantly different ($p<0.05$).

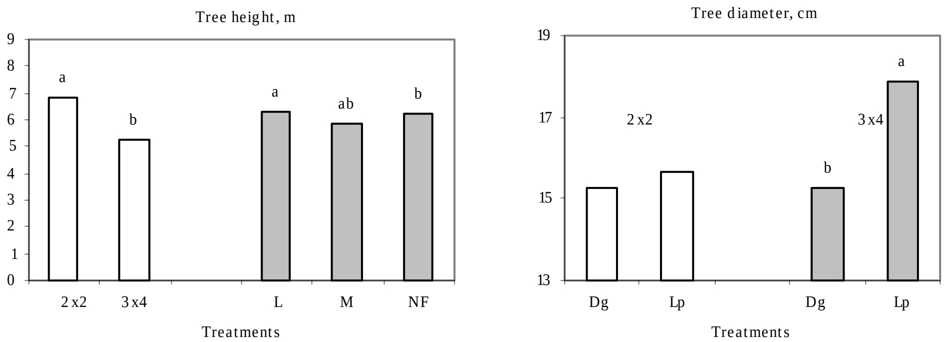


Figure 2. Tree height and diameter per density (2x2:2500 trees/ha and 3x4:833 trees/ha) and fertilization (L:milk sewage sludge; M:mineral fertilization and NF:No fertilization) and mixture (Dg:coocksfoot; Lp:ryegrass) treatment. Different letters indicate significant differences between densities or mineral fertilisation.

The botanical composition dynamics of the species sown can be seen in Figure 3 and depended on the sown mixture and fertilisation treatment. Ryegrass persistence was very low (maximum 5%) and better when mineral fertilisation was applied, at a low density and when ryegrass was sown. Cocksfoot reached fifty percent when it was sown previously in the first cut.

The *Agrostis* and *Holcus* percentages can be seen in Figure 4, which shows that these species had a higher presence when ryegrass was sown previously, *Agrostis* having a higher presence in the spring and *Holcus* during the autumn.

There were important differences between treatments with respect to pinewood percentage as it was significantly higher at a high density (between 25 and 55%) than at a low density (between 0.811 and 10%). That makes pasture even less productive at a high density, as sward was mainly made up of dead pinewood. The higher pinewood percentage of those plots sown with an Lp mixture at a 3 x 4 density can be related to a higher level of tree growth.

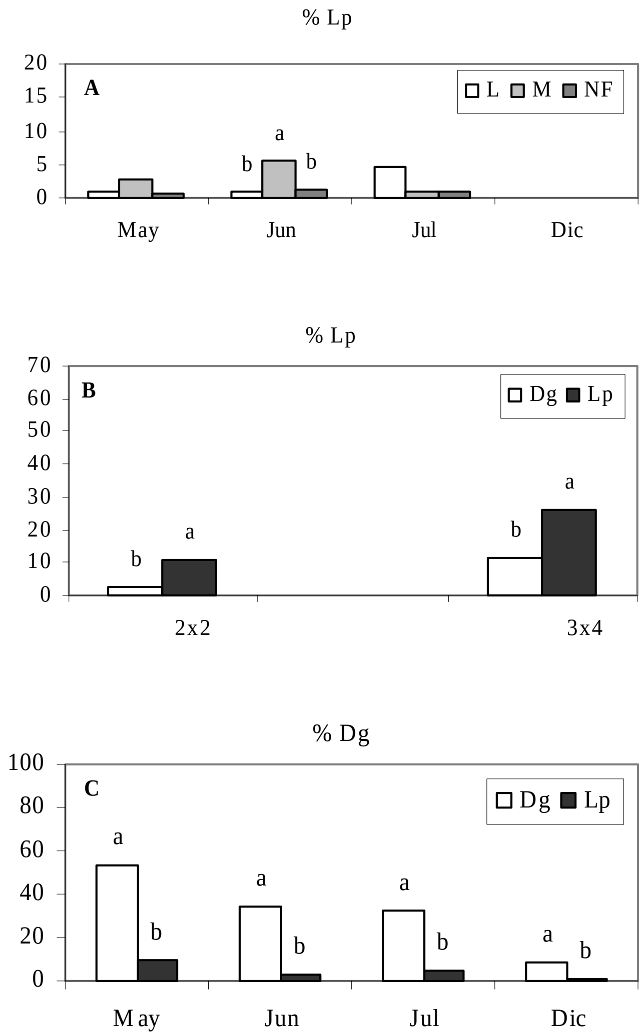
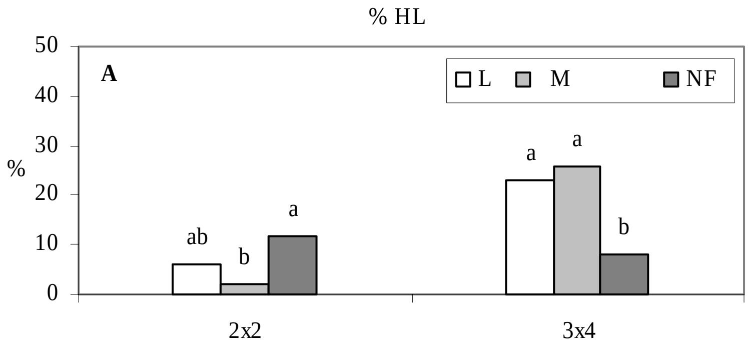


Figure 3. Percentage of sown mixture species (%Lp:ryegrassM %Dg:Cocksfoot) in the different treatments (Dg: Dactylis mixture sown and Lp: Lolium mixture sown; 2x2:2500 trees ha^{-1}; 3x4:833 trees ha^{-1}) a) differences between fertiliser treatments in each harvest; b) differences between Dg and Lp sownm misxture treatments in each density and c) differences between Dg and Lp sown mixture in each harvest.

At a high density when cocksfoot was sown pinewood percentage was increased in no fertilisation treatment and replaced by cocksfoot in fertilised treatments (Figure 3 and 5).

Yorkshire fog was increased at a high density when no fertilisation was applied but the opposite occurred at a low density (Figure 4). Bentgrass was reduced in mineral treatment, being replaced by Yorkshire fog at a low density.

Tree growth modifies soil characteristics such as the pH. The pinewood proportion can be associated negatively with the pH. On open swards mineral fertilisation usually reduces pH significantly, due to nitrate leaching and pasture extraction, but when a *Pinus* silvopastoral



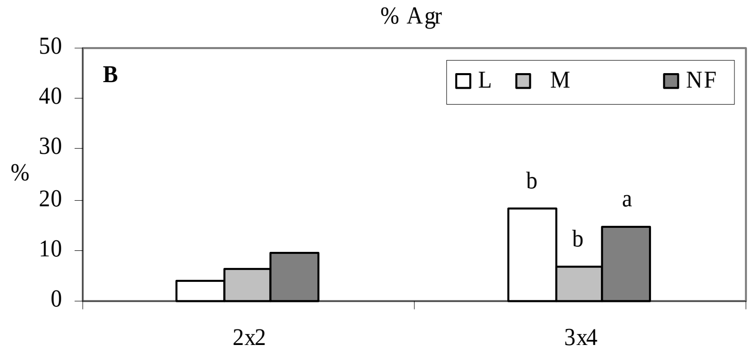


Figure 4. Percentage of *Holcus lanatus* (Hl) and *Agrostis* (Agr) into each fertilisation treatment (L:milk sewage sludge; M:mineral fertilization and NF:No fertilization) and density (2x2:2500 trees ha^{-1} and 3x4:833 trees ha^{-1}) Means within density followed by different letter are significantly different ($p<0.05$).

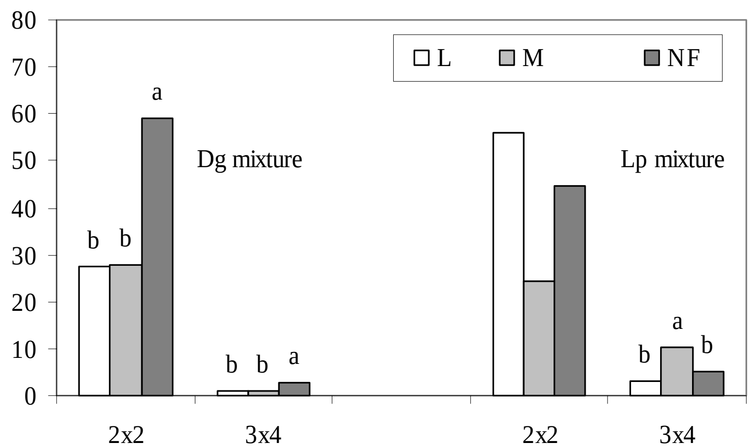


Figure 5. Pinewood percentage per density (2x2:2500 trees ha^{-1} and 3x4:833 trees ha^{-1}) and fertilization (L:milk sewage sludge; M:mineral fertilization and NF:No fertilization) treatment. Means within density and mixture followed by different letter are significantly different ($p<0.05$).

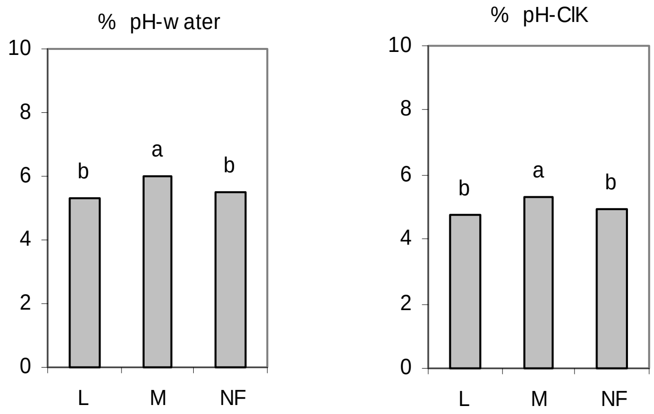


Figure 6. Water and ClK pH in each fertilised treatment ((L:milk sewage sludge; M:mineral fertilization and NF:No fertilization) Means within density followed by different letter are significantly different ($p<0.05$).

system is established and treatments caused different tree growth, those plots with larger trees, and therefore with a higher level of litter fall (pinewood), brought about a low pH, due to the pinewood acid effect.

Productive components also interact, as the pasture production response depends on tree density; if light did not reach soil no fertilisation response is found. Tree characteristics are also affected by density, the trees which have grown at a high density being taller and thinner. With plantations better spaced, the diameter depended on the mixture sown, being larger when *Lolium* was sown, due to the poor persistence of this species, that is replaced by others such as *Holcus* or *Agrostis*, that grows usually in unfertile conditions and generates less competitiveness with trees than *Dactylis*.

Experiments with silvopastoral systems with higher tree density indicated the lower limit of pasture production response to fertilisation due to a low input of light.

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Responses of Main Shrub Species to Different Grazing Regimes in Galicia

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Abstract

The objective of the experiment was to evaluate the effect of two grazing systems (rotational and continuous) on crude protein and pasture production under a 25-year old plantation of *Pinus radiata* (800 trees/ha) Horse grazing reduces pasture production and therefore the risk of fire. Differences in pasture production between treatments were not important, which makes a continuous grazing system more recommendable. Species' persistence depends on the horses' preferences. They preferred gorses instead of fern or bramble. Differences in pasture protein percentages were not important, with the exception of *Pteridium*, which had higher protein content when lamina percentage was higher.

Keywords: Silvopastoral systems; Pinus radiata D. Don; grazing systems.

1. Introduction

Galicia is one of the regions with the highest forest productivity potential in Europe. Sustainable forest management is an important goal for the EU agrarian policies, and it is accepted that it is a way of managing forests while providing multiple benefits, including wood and non-wood production, as well as soil protection, conservation of water and biodiversity, prevention of natural hazards, mitigation of climate change, social welfare and economic development (Mårell et al. 2003). Agro-forestry systems, as well as silvopastoral systems can be included in the objectives of sustainable and multi-purpose forest management as non-wood products can be obtained in the short and medium-terms, including beef and meat production from cows, goats or sheep, as well as mushrooms (favoured by grazing and faeces deposits). They are also systems

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that prevent natural hazards such as fires, the main problem Galician forests have to persist (the number of fires was over 55 000 between 1968 and 1989, meaning around 40% of the total Galicia area (3 mill. ha) and 60% of forestland. Only in 1989, 200 000 ha were destroyed by fire (Bermúdez-Alvite and Touza-Vázquez 2000).

Ensuring forest persistence will give sense to the important investments made in afforestation in the region (close to 20000 ha afforested in 2001 (Mosquera et al. 2001b)) and 400 000 ha in the last 12 years (Mosquera et al. 2001a), which will be lost if prevention measures are not taken. Prevention usually takes place by means of keeping a close watch on forests, and 10% of the budget is spent to this end. The best prevention is to destroy vegetal fuel for fire which has grown under the trees. Mechanical clearing cannot take place because it is expensive and no economical return is obtained, but animal clearing through grazing will convert fuel in food, generating rent from forests. Grazing can take place by means of employing different systems, such as continuous or rotational systems. Knowledge of the results of these two grazing systems and how they can control vegetation and productivity will help to improve management. Horses are known prefer some shrub species like *Ulex* instead of *Rubus* which are choosen with preference by goats (Mosquera et al. 2001). *Pteridium* has been described as toxic for some lignivorus animals, but horses eat them in some extent.

Management of grasslands mainly made up of grasses and herbs is well known in the region, and the main results indicate that rotational grazing will increase productivity in spite of making management difficult. However, the effect of different grazing systems on pasture productivity mainly made up of shrubs is not very well known. The objective of this study is to evaluate the effect of different grazing systems (continuous and rotational) with horses on *Ulex europaeus*, *Pteridium aquilinum* and *Rubus* sp development.

2. Material and Methods

The experiment started in 2000 in San Breixo, Parga (NW Spain). The experimental design was a randomised block with two replicates installed under a plantation of 25-year-old *Pinus radiata* with a density of 800 trees/hectare. Two treatments were established at a global stocking rate of 0.33 horses/ha. The continuous system treatment consisted of free grazing on the whole plot and the rotational system treatment consisted of sequentially grazing each sub-plot of the four established subplots within the main plot of 6 hectares. Every sub-plot was grazed on for 1 month, with a rest period of three months, which meant a instantaneous stocking rate of 1.32 horses/ha. In the experimental area the understory layers were mainly occupied by *Ulex* (90%), but species like *Pteridium aquilinum* and *Rubus* sp. were also in the test. Sampling took place in every vegetation type, that is, *Ulex*, *Pteridium* and *Rubus*, by cutting three randomised samples of 1 square meter before and after grazing at an height of 5 cm. Exclusion cages were used for estimating pasture growth under grazing. Offer biomass pasture was estimated by the sum of the pasture before grazing plus pasture growth during the grazing month under the rotational system and by the sum of the pasture outside the exclusion cages plus pasture growth (during the same grazing month as the rotational system) under the continuous system (Campbell 1966).

Leaves and woody twig fraction less than 0.5 cm in one case and woody twig fraction greater than 0.5 cm were handly separated in the laboratory for dry matter production estimation (48 h at 60 ºC), mowed and digested with the Micro-kjheldal technique for crude protein analyses (Castro et al. 1989) with TRAACS 800+ autoanalyser. ANOVA, taken in account blocks and treatments as factors, was used for the statistical analyses.

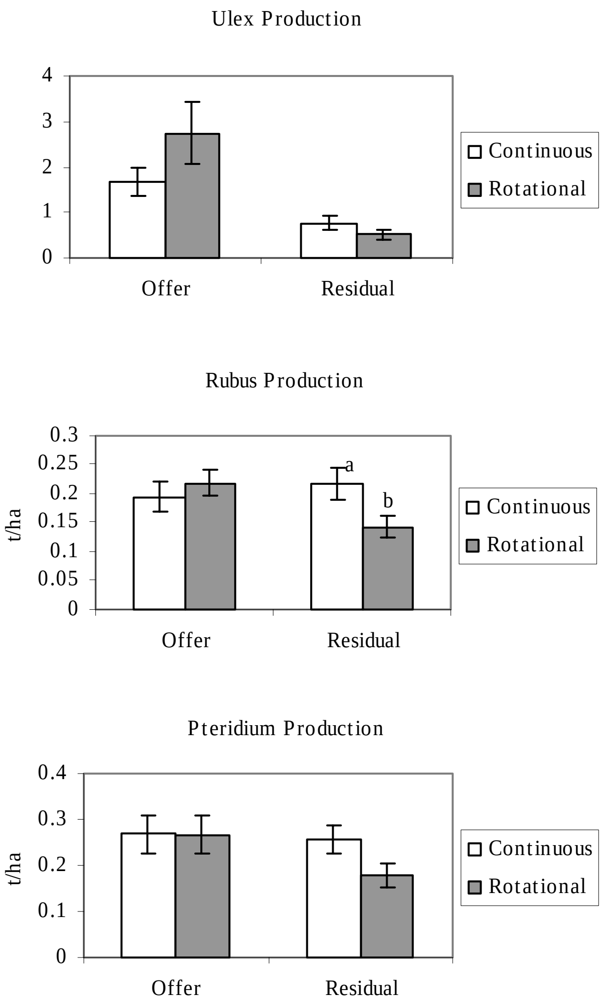

Figure 1. Offer and Residual pasture production of *Ulex, Rubus* and *Pteridium* in each treatment. Letter different means significant differences between treatments ($p<0.05$). t/ha:tons per hectare.

3. Results and Discussion

Ulex offer pasture was higher under rotational management than under the continuous grazing system (significant at a 10% level), however no differences were found for offer pasture of *Rubus* and *Pteridium* between the two grazing systems studied (Figure 1). The explanation for this could be because in the first rotation period horses had access to the whole area under continuous management, but every time that offer pasture was measured in the rotational plots during the first rotation, they had not been grazed on previously. Absence of a response from *Rubus* and *Pteridium* to grazing management can be explained as they were not preferably eaten by horses, which prefer gorse.

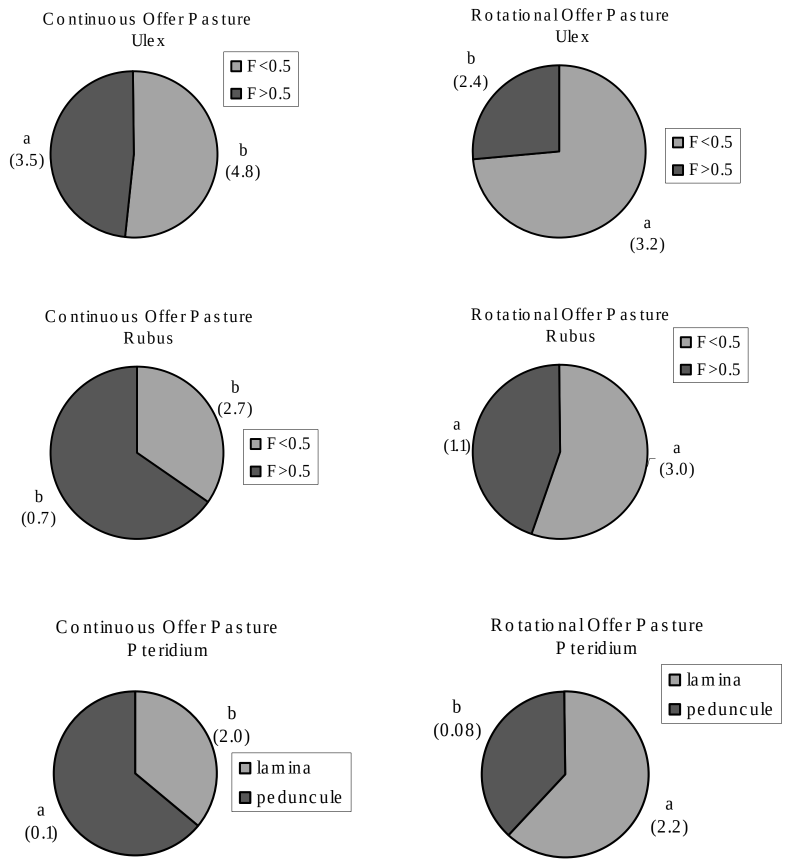

Figure 2. Offer pasture fractions (%) of *Ulex*, *Rubus* and *Pteridium* in each treatment. Letter different means significant differences between treatments (p<0.05) number between brackets is the standard error. F:Fraction.

On the other hand, when residual pasture is evaluated (Figure 1), there were no differences for *Ulex* species between grazing systems, as *Ulex* was eaten preferably by horses under both grazing systems. However, *Ulex* residual pasture tend to be lower under rotational systems, this can also be seen with fern and bramble. The different response of the three species studied to grazing management can be explained by the selection capability of horses for the different species. They prefer gorse, which caused an offer pasture reduction, but when there is not enough food in the plots, which equals gorse residual pasture, then they ate bramble, preferably, or fern, as can be seen for residual pasture of bramble.

The percentage of fractions with a diameter of branches lower and higher than 0.5 cm for offer pasture of *Ulex* and *Rubus* and for the stem and lamina of *Pteridium* in each treatment can be seen in Figure 2. A more desirable fraction of *Ulex*, *Rubus* and *Pteridium* species, that is to say fractions lower than 0.5 cm or lamina, were always higher under the rotational than under the continuous grazing system, due to the fact that there was no grazing under the rotational treatment in the first rotation when offer pasture was estimated.

When the percentage of fractions was evaluated in the residual pasture, the relationship between the two fractions in the two treatments was different to that of the offer pasture

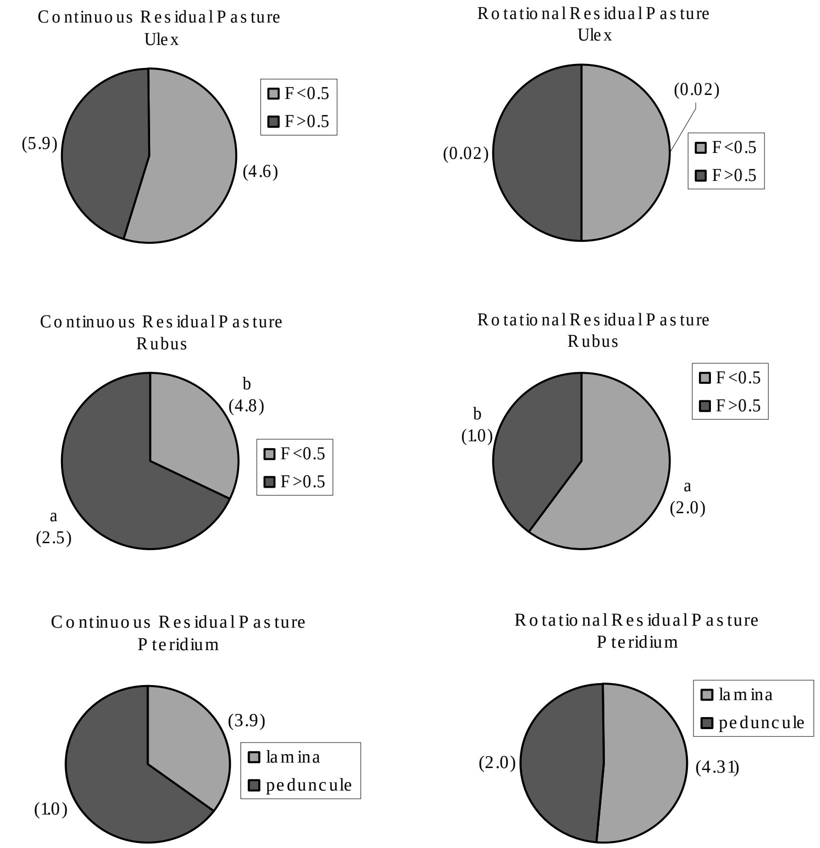

Figure 3. Residual pasture fractions (%) of *Ulex, Rubus* and *Pteridium* in each treatment. Letter different means significant differences between treatments (p<0.05) number between brackets is the standard error. F:Fraction.

(Figure 3). Most of them reduced the preferred pasture fraction, with respect to the offer pasture under the rotational treatment, but relative proportion percentages were maintained under the continuous system. *Ulex* had a higher percentage of small fractions in continuous grazing as compared to rotational grazing, as this is the species preferred by horses and it is eaten vigorously, and some growth is allowed under the continuous system. However, the opposite occurred in *Rubus* and *Pteridium*, as the percentage of buds was lower under the continuous treatment. That can be explained because horses chose buds under the continuous system, but there was no selection capability in the rotational system where leaves from widest fraction and stems were consumed due to the highest instantaneous stocking rate, and this meant that there was no difference in the percentage between offer and residual pasture for rotational treatment for both species. The differences between the treatments are mainly explained by stocking rate and horse food preferences, as happened with *Rubus* and *Pteridium*, as they had less production.

Protein concentration in pasture in each fraction and treatment can be seen in Figure 4. Protein content is higher in the offer pasture of *Ulex* and *Pteridium* than in *Rubus*, as legume species usually have more protein concentration, as the former is a legume and fern usually

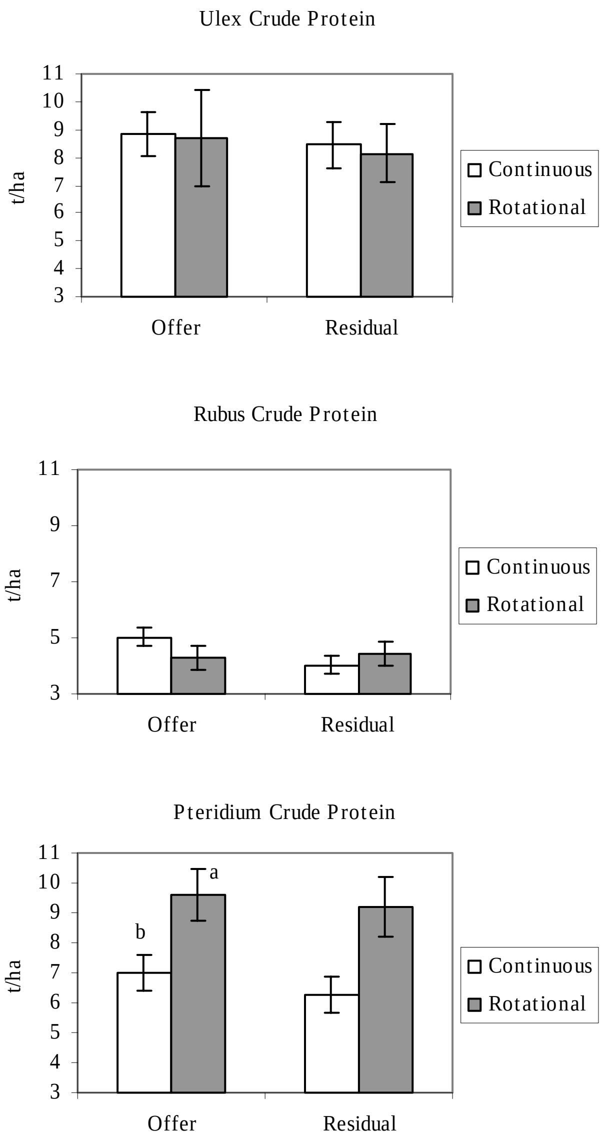

Figure 4. Offer and Residual crude protein (%) of *Ulex*, *Rubus* and *Pteridium* in each treatment. Letter different means significant differences between treatments ($p<0.05$).

grows in more fertile sites. Percentage values varied between 8 and 9 for *Ulex*, 4 and 5 for *Rubus* and 6 and 10 for *Pteridium*. There were no differences in protein content between treatments for *Ulex* and *Rubus*. *Pteridium* had the best protein content under the rotational system for offer pasture, and the same tendency was found for residual pasture. This could be explained because lamina had higher protein content than stem, fern offer pasture production was similar between treatments and the differences between lamina fractions were very important in offer pasture, as this percentage was 35% and 61% for the continuous and rotational systems, respectively.

Horse grazing reduces pasture production and therefore the risk of fire, which is an important sustainable aspect in forest systems based on gorses. Differences in pasture production between treatments after the first rotation were not important, which makes it more recommendable to use a continuous grazing system, as it is cheaper due to the cost reduction in fencing. Species persistence depends on horse preferences. Less preferred species like fern or bramble are consumed less in the continuous system due to its low instantaneous stocking rate, which allows horses to eat more palatable fractions and can increase the persistence of this species, and therefore maintain biodiversity better than the rotational system. Differences in pasture protein percentages were not important, with the exception of *Pteridium*, which had a higher protein content when the lamina percentage was higher.

Acknowledgements

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Pinus radiata D. Don growth and pasture production in fertilised silvopastoral systems in Galicia, NW Spain

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Abstract

The object of this test was to study the residual effect of liming, as well as the effect of applying an organic fertiliser (sewage sludge) and a non-organic fertiliser in a silvopastoral system characterised by an acid soil (pH=4,5) and low in nutrients. The study took place in Lugo (NW Spain) at an altitude of 510 m, in plots where two doses of lime (0 and 2.5 t ha^{-1}) were applied and which were fertilised organically (0, 160, 320 y 480 t N-total ha^{-1}) and inorganically (500 kg 8:24:16 ha^{-1}) over a period of four years (1997–2000), a non-organic fertiliser of 250 and 500 kg 8:24:16 ha^{-1}respectively being applied in 2001. The use of sewage sledge brings with it an increase in pasture production, as well as increasing the height and diameter of the trees.

Keywords:Sewage sludge; silvopastoral system; Pinus radiata.

1. Introduction

There is an important demand for wood within the European Union today, which, along with the closure of numerous dairy farms, in Galicia is favouring reforestation of areas of shrubland and the re-conversion of agricultural land into forestry plantations. The increase in the surface area with tree cover and the subsequent risk of forest fires lead to an increase in the budget given over to fire prevention.

A management option which would improve the profitability of the system, by diversifying production and reducing fire prevention costs, would be the joint use of forestry and farming systems, combining high-quality wood production and the understorey growth control by allowing animal grazing.

When installing silvopastoral systems it is necessary to bear very much in mind pasture production, which must cover the feeding needs of the animals which are to graze there. In

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Galicia, mountain soils are in general acid and have low fertility values (Muñoz-Taboadela 1965), which creates the need for fertilisation and liming in order to favour their productivity. In recent times, the use of organic fertilisers to improve pasture production is being promoted.

Also, as the construction of sewage treatment plants in areas with low population densities is obligatory (Guideline 91/271/EEC), an increase in the production of organic waste, such as urban sewage sludge, is expected.

In Europe the most common waste water treatment that takes place is biological, which implies high energy consumption and generates important quantities of sludge (biodegradable waste).

To date, the main methods used in order to eliminate such waste, in Spain as well as in Europe were to accumulate it in dumps (whose fill-up level accelerates) recycle it, incinerate it (a high-cost process) and dump it at sea (prohibited in Spain since 1999).

Therefore, an interesting option is the use of this biodegradable waste in agriculture, as fertiliser, as in this way nutrients, such as nitrogen, phosphorus and potassium would be re-used, and at the same time they would be re-valued from the point of view of their use in agriculture, as indicated in European Directive 86/278/EEC, which also points out that in the face of the use in agriculture of sludge it is necessary to bear in mind the needs of plants, so as not to negatively affect the soil and its productivity, making it essential to carry out sampling and analyses of the substrata which this kind of fertiliser has been applied to for reasons of environmental safety.

It must be taken into account that different factors involved in obtaining a fertiliser influence the fertilising quality of the sludge, among which we would underline the nature of the effluent or origin of the residual waste water which reach the treatment plant, the procedures used for treating the waste water and the treatments the sludge undergoes.

The continued application to the soil of fresh sludge, damp on the surface can favour the formation of a surface layer which might make it difficult for plants to develop. In these cases, it would be advisable to apply, after the first phases of organic fertilisation, mineral fertilisation to the surface, which would accelerate the mineralization of the mud, reducing the persistence of said layer. Another solution, if such a circumstance were foreseen, could be to reserve the mud for applications mixed with soil when the crops (pasture land in our case) are being installed and to fertilise the surface with mineral fertilisers.

2. Material and methods

The experiment took place at a site in Lugo (NW Spain) at an altitude of 510m. Average annual rainfall is 1350 mm/year^{-1}. The study took place in 2001 and was based on a test established in 1997.

Therefore, in 1997 the scrubland existing in the lanes that define the lines of woodland in a grove of five-year old *Pinus radiata* D. Don and with a density of 1667 trees ha^{-1} (3 x 2 m) was cleared. The elementary plot installed measured 96 square metres (12 x 8 m). The total number of plots in the test was 27 (9 treatments and 3 replicas), in 12 of which liming took place with a dose of 2.5 t/ha of $CaCO_3$. Following that, in all the plots fertilisation took place, using 120 kg/ha of P_2O_5 and 200 kg/ha of K_2O, after which 25 kg of *Lolium perenne* cv Brigantia, 10 kg of *Dactylis glomerata* cv Artabro and 4 kg of *Trifolium repens* cv Huia were sown per hectare.

Both in the plots limed as well as in those not limed, different surface maintenance fertilisation treatments were established, consisting of the application of 4 doses of sludge with which the following quantities of nitrogen were incorporated into the soil: 0 (NF), 160 (LB), 320 (LM) and 480 (LA) kg N-total/ha, of which it is assumed approximately a fourth is assimilated (EPA 1994). In the case of the plots not limed an additional treatment was installed (MIN) consisting of the application of 500 kg/ha of the complex 8:24:16, equivalent to the fertilisation applied traditionally in the area. The fertilisation treatments used in 1997 were repeated annually on the surface up until 2000, being applied in plots of 96 m^2 delimited by 25 trees. The design used was completely random with three replicas.

The sludge used in the fertilisation process came from the waste water plant in Lugo (GESTAGUA, S.A.), and showed apt characteristics for use as an organic fertiliser.

In April 2001 non-organic fertilisation took place using 250 kg/ha of the fertiliser complex 8:24:16 in the plots which had been fertilised with sewage sludge, applying a dose of 500 kg/ha of the same fertiliser in the mineral fertilisation treatment.

Subsequently, pasture samples were taken in May, June and December 2001, providing an estimation of the total production of pasture and the botanical composition (in each plot 4 samples of 0.09 square metres were taken at random and dried at 60ºC for 48 hours).

In December 2001 soil samples were taken at two depths (5.5 y 25 cm, in accordance with Spanish legislation, RD. 1340/90). The samples were dried in the air and passed through a 2 mm mesh and subsequently the pH in water was determined (2.5:1).

In the case of the woodland, in August 2001 height and diameter measurements were taken for the trees located inside the plots in order to avoid the boundary effect.

The results obtained were assessed using ANOVA and the averages were separated using the Duncan test; the statistical pack SAS was used for all of the process.

3. Results

The data obtained insofar as pasture production was concerned indicates that the total production fluctuates between 2.9 and 5.6 t DM/ha (Figure 1). Of the species sown, *Dactylis glomerata* is the one that contributes the greatest, as it adapts well to shade and to soils with a low level of fertility (Piñeiro and Pérez 1990).

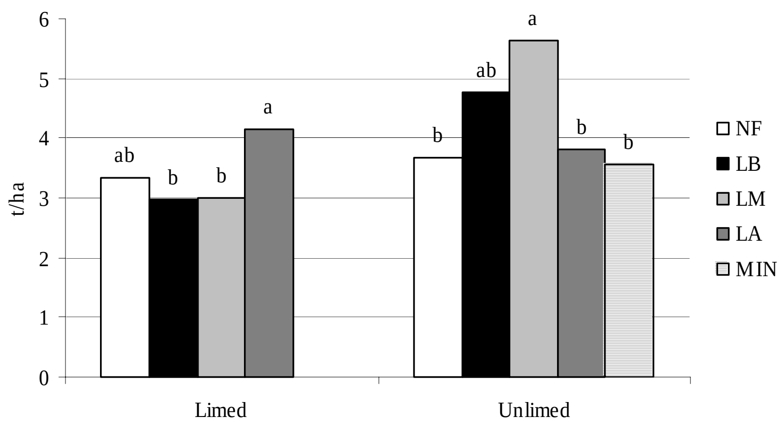

Figure 1. Annual pasture production for each fertilised treatment (NF: no fertilised; LB: low sewage sludge dose; LA: High sewage sludge dose; Min: mineral) in limed and unlimed plots. Different letters indicate the existence of significant differences between the fertilisation treatments carried out.

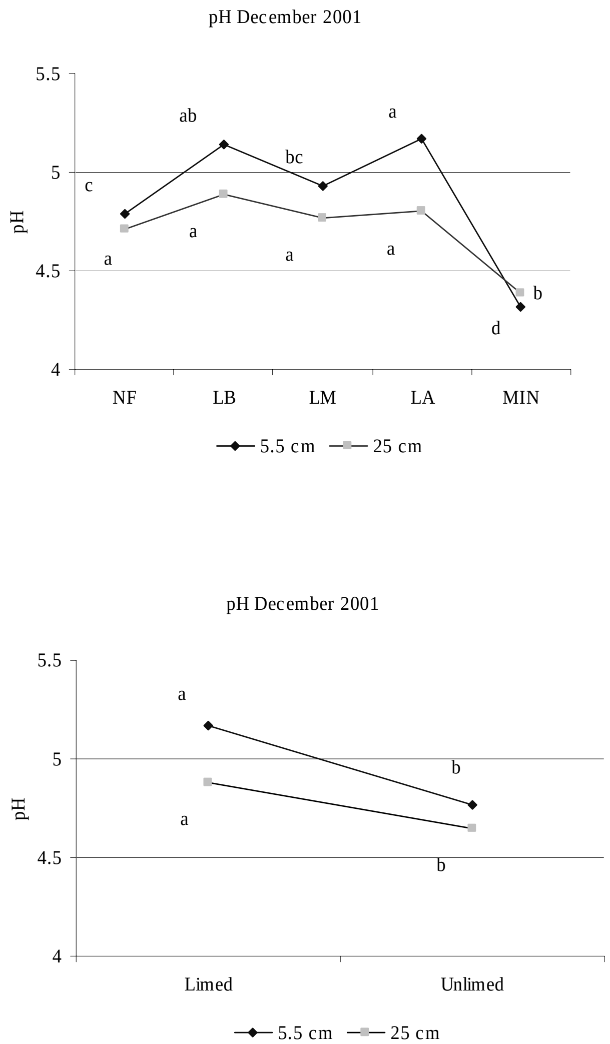

Figure 2. Water pH per treatment (NF: no fertilised; LB: low sewage sludge dose; LA: High sewage sludge dose; Min: mineral) in both depth. Different letters indicate the existence of significant differences between the fertilisation treatments carried out.

The total production comes within the range quoted for mountainous terrain based on other experiments carried out in Galicia (Mosquera-Losada et al. 1999; López-Díaz et al. 1999).

The highest levels of pasture production can be associated with high doses of sludge, when lime was applied, while, when lime was not applied, higher levels of production are obtained with low and medium doses of sludge.

The positive response in the plots not treated with lime can be explained by the fact that sludge produces an increase of soil pH (Figure 2), the same effect that liming.

Results from other experiments (López-Díaz et al. 2001) also indicate that higher production levels can be associated with high doses of sludge, in comparison with those registered in the

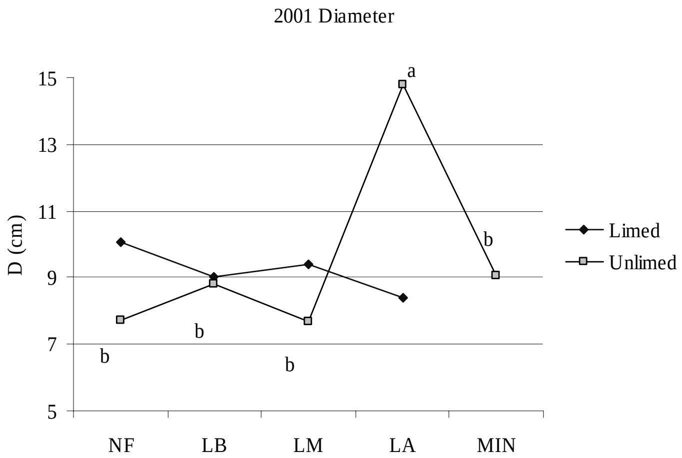

Figure 3. Tree diameter for each fertiliser treatment (NF: no fertilised; LB: low sewage sludge dose; LA: High sewage sludge dose; Min: mineral) in limed and no limed plots. Different letters indicate the existence of significant differences between the fertilisation treatments carried out.

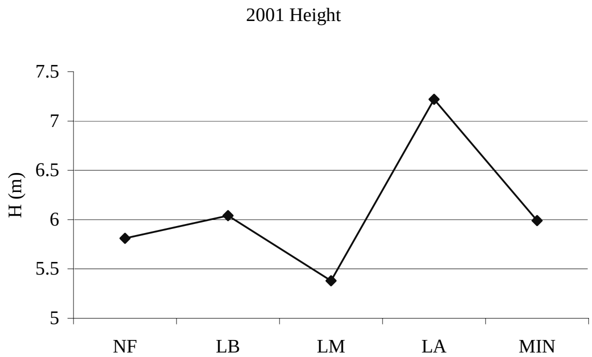

Figure 4. Tree height for each fertiliser treatment (NF: no fertilised; LB: low sewage sludge dose; LA: High sewage sludge dose; Min: mineral).

NF treatment, although in our case the differences are not significant on account of the difficulty of incorporating the sludge applied in high doses after four applications when liming does not take place.

Insofar as the woodland is concerned, it was observed (Figure 3) in that non-limed plots, where the LA dose was applied, trees had a significantly larger diameter than those that received the other doses, a behaviour similar to that detected in other experiments (Rigueiro et al. 2000), López-Díaz et al. 2001).

When liming does not take place, the lower height (Figure 4) and diameter of the trees with the LM dose is explained by the fact that with this treatment pasture production was higher than that registered with the others, which would lead us to conclude that in this case the pasture assimilates the nutrients better, as Clinton and Mead (1994) and Rigueiro et al. (2000) also pointed out, a greater competition being established between the tree species and the different pasture species.

Organic fertilisation produces a more positive effect than mineral fertilisation on pasture production and the growth of *Pinus radiata,* when lime is not applied, as the application of sewage sludge raises the level of pH in the soil while mineral fertilisation acidifies it.

The differences between the effects of the different fertilisation treatments are reduced when a liming agent is applied.

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Towards a Centre for European Forest Science

Towards the Sustainable Use of Europe's Forests: a Centre for European Forest Science! – Some Missing Catalytic Elements?

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Among the objectives of the Symposium in Tours was the identification of important scientific topics to be part of the future NoE, and of the partners having the capacity to implement the corresponding research. The present paper is a contribution in this direction.

Abstract

The Symposium, *"Towards the sustainable use of Europe's forests"*, with sub-title *"Forest ecosystem and landscape research: scientific challenges and opportunities"* lists three fundamental substantive areas of research that are involved: *Forest management and practices, Ecosystem processes and functional ecology, and Environmental economics and sociology*. This paper argues that there are essential catalytic elements missing! Without these elements there is great danger that the aimed-for world leadership in the forest sciences will not materialize.

What are the missing elements? All the sciences, and in particular biology, environmental sciences, sociology, economics, and forestry have evolved so that they include good scientific methodology. Good methodology is imperative in both the design and analysis of research studies, the management of research data, and in the interpretation of research finding. The methodological disciplines of Statistics, Modelling and Informatics ("SMI") are crucial elements in a proposed Centre of European Forest Science, and the full involvement of professionals in these methodological disciplines is needed if the research of the Centre is to be world-class.

Distributed **V**irtual **I**nstitute (DVI) for **S**tatistics, **M**odelling and **I**nformatics in **F**orestry and the **E**nvironment (SMIFE) is a consortium with the aim of providing world-class methodological support and collaboration to European research in the areas of Forestry and

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the Environment. It is suggested that DVI: SMIFE should be a formal partner in the proposed Centre for European Forest Science.

Keywords: research methodology, statistics, modelling, informatics, DVI: SMIFE.

Introduction: What is Forest Science?

Forests are probably the world's main renewable natural resource, and a vast repository of the world's biodiversity resources. Forestry is an activity that spans the world, so that efficient and effective management of the forestry enterprise for the sustainable production of a wide range of products, from timber to medicinal plants, from sociological to ecological environments, for both human and animal populations is of crucial importance. See SCOPE (Scientific Committee on problems of the Environment), "An interdisciplinary body of natural and social science expertise focused on global environmental issues, operating at the interface between scientific and decision-making instances", for some relevant publications.

However, efficient and effective sustainable management of forest resources depends on a scientific understanding of the structure and processes of forests, of the dynamics of their interactions with other populations and processes in the world ecosystem. The science of such complex and hierarchical systems is not straightforward, of course, (Ahl and Allen 1996). Additionally, objective measurement, assessment and monitoring methods are necessary to determine the current state of forest resources. Together, understanding of structure and process, and knowledge of the current state, allows prediction of future developments under a range of alternative future management practices. A quantitative scientific approach is essential to successfully maintain control of the sustainable management of forest resources.

But what is the "scientific approach"? There are many scientific disciplines which have direct relevance to forestry and environmental research: biology and botany, chemistry and biochemistry, physics and engineering, ecology and systems research, meteorology and climatology, geology and Geographic Information Sciences, sociology and economics, … to mention but a few. Does the "scientific approach" within forestry research amount to the use of the accumulated knowledge and understanding of these underpinning scientific disciplines? Certainly, this must be a part of what "Forest Science" is about. However, it is not all!

The "scientific approach" is not just about the use of existing scientific knowledge and understanding. It is also about the methodology by which new scientific facts and understanding may be discovered. Such research methodology is a feature shared by all the substantive scientific disciplines mentioned in the last paragraph. First, such research methodology is based on the "scientific method" which is the basis of all modern science: observation (data-collection), hypothesis/conjecture, prediction test (designed experiments), theories and models, (Locke 1690).

The application of the Locke's "scientific method" in the disciplines of biology, ecology, the environment, forestry, sociology, and economics was found to be particularly problematic in the 18th and 19th centuries. For example, when Professors Jones, Quetelet, Malthus, and Babbage proposed a new section of the British Association, in Cambridge, in 1833, but the considered response was that the "things with which the Association had to do were the laws and properties of matter and with those alone", a response leading to the formation of the London Statistical Society one year latter. The systems under study in these disciplines outside the physical sciences were highly complex and dynamic, with many component processes, and much variably in the observed data. The classical methods for the use of the scientific experimentation in physics and engineering, i.e. keep all conditions fixed except one controlling variable, and observe the output as that controlling variable is varied, were not applicable. Most variables were not individually controllable, most of them interacted with others, and the nature of the systems meant that the observed data was so variable, due to uncontrolled factors, that there were great problems in identifying the patterns and relationships underlying the observed data. So was born, early in the 19th century, from the methodological challenges of the non-physical sciences, the methodological discipline of **Statistics**: specifically concerned with identifying patterns and relationships in observational and experimental data that was "messy", and assessing the significance of such patterns in the context of high variability in the observed data.

"As Schrödinger (1949:91) has remarked, 'I dare say the first scientific man aware of the vital role of statistics was Charles Darwin'".

M.S. Bartlett (1962).

Hence, **Statistics** is an essential catalytic methodological discipline essential for modern research in forestry, the environments, etc…. As one of the founder-fathers of modern (mathematical) Statistics said , (more than a century after the BA (1833) meeting):

"To call in the statistician after the experiment is done may be no more than asking him to perform a post-mortem examination: he may be able to say what the experiment died of."

Ronald A. Fisher. Indian Statistical Congress, Sankhya (ca 1938)

However, there are worrying signs that the essential role of Statistics might be overlooked:

"I am convinced that we, as statisticians, have largely failed to get even the most basic ideas across to our colleagues from other disciplines. What is it that we can do to persuade people to embrace these exciting ideas and use them in their research and in everyday management?"

John Jeffers, FBMIS (2003)

Modelling in Forest Research

"It is widely assumed - particularly by statisticians - that the only branch of mathematics necessary for a biologist is statistics."

John Maynard Smith (1968)

The construction of models to represent the patterns observed in observational and experimental data is a core element of the scientific process, and hence all scientists are to some extent "modellers". Models take many forms: conceptual and computational, deterministic and stochastic, and in most modern scientific research all forms of model feature. There are many families of models, each having their own properties, and these will often determine if a particular model is appropriate or not for use in a particular situation or research study, (Doucet and Sloep 1992; Brown and Rothery 1993). It is generally accepted by modellers that "there is no perfect model": modelling is the process of continual refinement of models, within the cycle of the scientific method, so that the models/theories progressively improve in descriptive and predictive power.

"Roughly speaking, a model is a peculiar blend of fact and fantasy, of truth, half-truth and falsehood. … It is … just as great a mistake to take the imperfections of our models too seriously as it is to ignore them altogether …"

J.G. Skellam (1973)

However, sometimes a given modelling paradigm reaches the end of its potential scientific use, and in such a situation it is necessary to develop new model frameworks, based upon previous models, but going beyond them.

Both the incremental development of models within a given modelling framework, and the development of new modelling frameworks requires extensive knowledge and experience of the modelling frameworks available, their strengths and limitations, and the ability to harness such model frameworks to the scientific discipline concerned and the unique features of that discipline's data.

> *"Mathematics gained entry into the natural Sciences by the Statistical door, but this stage passes to the stage of analysis as in all qualitative sciences."*
>
> *Kostitzin (1939), cited by Bartlett (1962).*

While all scientists are modellers, "Modelling" is also a methodological discipline in its own right. Naturally, the depth of knowledge and facility of in modelling methodology varies considerably amongst substantive scientists. Some have become experts in such modelling techniques (e.g. Turing 1952); some are not so expert, relatively speaking. However, in any new strategic initiative to develop new scientific programmes, such as the setting up of a new Centre for European Forest Science, it is important for the quality of the future forest science produced that professional modellers should be involved in the enterprise, as well as the substantive scientists in biology, ecology, forestry, sociology, etc….

Informatics in Forest Research

Information, knowledge and understanding are the target aims of all scientific research. Models are a means of capturing and representing in a concise and useable form this information, knowledge and understanding. Statistics enables efficient and effective data-collection studies and experiments to be designed, analysed, and interpreted in the context of the scientific models under consideration.

The main modern means for achieving the statistical and modelling goals makes extensive use of computers for data collection and storage, data-management, design and analysis of experiments and surveys, implementation of models, and the construction of the Decision Support Systems that eventually make use of the underpinning scientific research. There are also many new methodological threads associated for such use of computers this research: information systems, software engineering, machine learning and A.I., the semantic web,… etc. Such is "Informatics". There is no need to make a case for Informatics as a crucial methodological discipline in modern research and management endeavours. It is generally accepted.

However, there is a case to be made for the full inclusion of specialists in Informatics in forest and environmental research activities and institutions. Computing and Informatics are often regarded as secondary support disciplines in the scientific research process. This is a mistake. Informatics is a crucial catalytic methodological discipline, along with Statistics and Modelling.

The Catalytic Methodological Elements

It has been argued above, (even though many would regard the arguments as self-evident), that **S**tatistics, **M**odelling and **I**nformatics (**SMI**) are crucial methodological disciplines that should be included fully in any modern scientific enterprise.

Unfortunately, partly because all scientists are modellers, to some degree, because most scientists will have had some basic exposure to statistical methods, and because all scientists made use of computers in their work, there is a tendency to ignore the need to fully include specialists in the SMI methodological disciples into early design phases of new scientific endeavours. To make such an omission is a major strategic error. The “reaction” will not run without its full dose of essential catalysts! But how should these catalysts be added to the reaction mix?

The need for Collaboration, Cooperation and Research in SMI in Forestry

The message of history of science is clear: SMI are disciplines that are central to the healthy development of research in Forestry and the Environmental Sciences. However the modern European scene is rather fractionated in the SMI area, and within the sciences to which they are essential there is a worrying blind-spot to the importance of SMI. Historically, I suppose this is not really surprising. The deliberations of the BA in 1833 were rather short-sighted. Fisher in 1938 still did a lot of post-mortems, and Jeffers’ 2003 cry of frustration tells us that the story continues.

DVI: SMIFE

A Distributed Virtual Institute for Statistics, Modelling and Informatics in Forestry and the Environment (DVI: SMIFE)[1] was formed as a virtual organisation in 2003. The DVI brings together leading European specialists in the SMI disciplines having extensive professional experience of applying them in the areas of Forestry and the Environment. The aim of the DVI is to provide a structured framework within which SMI support and collaboration can be provided to substantive scientists in the forest and environmental research areas. More formally, and in the context of current EU Framework 6 activities:

The purpose of DVI: SMIFE is to:

1. be able to join other NoE and IP proposals as a substantial and world class institute in the methodological sciences.
2. be the basis of a core team for a NoE or an IP to address some of the main generic methodological problems and issues, the solution to which will have wide applicability across many areas of application.

The remit of DVI: SMIFE[2] :

(i) to bring the generic techniques of SMI to bear on the research and management problems of forestry, the environment, and possibly agriculture.

1 Current membership of DVI: SMIFE includes: Prof. Keith Rennolls, University of Greenwich, U.K. (Coordinating Chair); Prof. Dr. Dieter Pelz, Professor and Director of the Abteilung Forstliche Biometrie, Faculty of Forestry/University of Freiburg, Germany. Prof. Erkki Tomppo, Finnish Forest Research Institute, Finland. Professor Bo Ranneby, Center of Biostochastics, SLU, Sweden. Prof. Dr. Joachim Saborowski, Institute of Forest Biometry and Informatics, University of Göttingen, Germany. Prof. Dr. Christoph Kleinn, Institutes of Forest Management and Yield Science, University of Göttingen, Germany. Dr. Hans Voss, Manager Spatial Decision Support, Fraunhofer Institut Autonome Intelligente Systeme, Germany.

2 For more details on the SMIFE concept, see: http://cms1.gre.ac.uk/research/cassm/smife For examples of SMIFE contributions, see: http://cms1.gre.ac.uk/conferences/iufro/proceedings .

(ii) to consider the use and development of generic techniques for the modelling, and analysis of hierarchical, spatial and temporal processes, simultaneously as well as separately.
(iii) to clarify the definition of concepts, metrics, indices etc... used to specify, measure and monitor major features of the developing forest/environment ecosystem.
(iv) to consider methodological and quality issues concerned with the measurement, experiment, assessment, survey, and inventory processes.
(v) to develop and apply targeted tools from AI and knowledge discovery methodology to the FE areas.
(v) to ensure modern information systems techniques are used; the semantic web, the grid, ...etc.

Related activities include a new e-journal, FMBIS[3], and a prototype Forest Model Archive (FMA)[4].

Concluding Suggestion

The Statistical, Mathematical Modelling and Informatics disciplines should be included in plans for the future of research into the sustainable management of Europe's forest resources, and all associated forest research activities.

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3 For a new free on-line journal on Forest Biometry, Modelling and Information Sciences (FBMIS), see: http://www.fbmis.info/
4 For a prototype Forest Model (FMA), see: http://www.forestmodelarchive.info/

Postscript

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After the Tours meeting, the preparatory work has been coordinated by a Core Group relying on three advisory teams and chaired by EFI and ECOFOR. More than a hundred Expressions of Interest were received by the end of September 2003 thus demonstrating a wide interest in the scientific community for joining this NoE.

The NoE initiative was presented to the stakeholders and the Commission representatives in the final workshop of EFI's IMACFORD project on 10 October 2003. At the same time it was learned that there will most likely be only one topic on forestry in the 3rd call, and the topic will be the forestry-wood chain (in which some rewording might be expected).

Even if no topic on the multifunctional forest management appears in the 3rd Call, it may come in the 4th Call or in the 7th Framework programme. This has led to the conclusion that the preparation of NoE should be continued but with a longer time frame. A four-page description of the initiative and the Expressions of Interest will be the interim result of the preparation work thus far.

The current topic of the initiative is 'Establishing a European Scientific Platform for structuring research on Multifunctional Forest Management and its Sustainability Impact Assessment' (ESP-FOR).

The aim of ESP-FOR is to contribute to the European Research Area by designing a multidisciplinary scientific platform to facilitate a long-term programme mainly based on existing research capacities in the participating institutions. The platform will address building elements for reference scenarios, harmonised data-sets and modelling approaches for studying multifunctional production possibilities of the forest ecosystems, and the needs of societies for the products.

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