



Mapping and Monitoring of Natural Areas in the Nordic Countries

**- Proceedings from the workshop,
November 1-3, 2002, Fuglsø, Denmark**

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- Proceedings from the workshop, November 1-3, 2002, Fuglsø, Denmark

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Nordic Environmental Co-operation

Environmental co-operation is aimed at contributing to the improvement of the environment and forestall problems in the Nordic countries as well as on the international scene. The co-operation is conducted by the Nordic Committee of Senior Officials for Environmental Affairs. The co-operation endeavours to advance joint aims for Action Plans and joint projects, exchange of information and assistance, e.g. to Eastern Europe, through the Nordic Environmental Finance Corporation (NEFCO).

Nordic co-operation

Nordic co-operation, one of the oldest and most wide-ranging regional partnerships in the world, involves Denmark, Finland, Iceland, Norway, Sweden, the Faroe Islands, Greenland and Åland. Co-operation reinforces the sense of Nordic community while respecting national differences and similarities, makes it possible to uphold Nordic interests in the world at large and promotes positive relations between neighbouring peoples.

Co-operation was formalised in 1952 when *the Nordic Council* was set up as a forum for parliamentarians and governments. The Helsinki Treaty of 1962 has formed the framework for Nordic partnership ever since. The *Nordic Council of Ministers* was set up in 1971 as the formal forum for co-operation between the governments of the Nordic countries and the political leadership of the autonomous areas, i.e. the Faroe Islands, Greenland and Åland.

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Preface

Although the Nordic countries have different obligations for biological conservation – some are e.g. EEC-members others not – we are facing the same basic challenges. That is:

What is to be mapped and how?
How do we characterise mapped land units?
What units should be monitored?
- and on what scale and by which parameters?
How do we transform monitoring data to meaningful indicators?
How do we set the objectives for the protected units?
How do we define favourable conservation status?
How do we distinguish between progress and regress in biological condition?

These questions motivated us to arrange a Nordic workshop, where scientists and administrators with insight and interests in the issues could meet and exchange knowledge and ideas. The workshop was established as an initiative funded by Nordic Council (NMD - Nordiska Arbetsgruppen för miljöövervakning och -Data, and NFK (Natur-, friluftslivs- og kulturmiljøgruppen). A parallel Nordic activity, reported in TemaNord 2001:523, regarded the broader issue of large-scale strategic landscape monitoring, whereas the workshop reported here was focussed on natural areas of high conservation priority.

The workshop, which took place in Denmark, November 1-3, was arranged by NERI and NINA in collaboration. It attracted participants from Norway, Finland, Iceland, Sweden, Denmark, Estonia and had invited speakers from the Netherlands.

All speakers at the workshop were given the opportunity to present their contribution for a wider audience in this proceeding. The contributed papers differ considerably in length, from short abstracts to extensive introductions or reviews. This simply reflects that not all speakers have had the opportunity to allocate the time required to produce a written contribution.

Workshop program

Mapping and monitoring of natural areas in the Nordic countries

Nordic workshop, November 1-3, 2002

Denmark, Fuglsøcentret.

Friday, November 1

16.00-20.00 Arrival and accommodation.

Saturday, November 2: "Value assessment and mapping of natural areas"

Introduction

09.00-09.15 Opening of the workshop

09.15-09.55 Conservation value in a Nordic context: Naturalness, ecosystem function and rarity - objectives and criteria. *Odd Stabbetorp NINA-NIKU, Norway & Harald Bratli, NIJOS, Norway*

Applications

09.55-10.30 Municipality mapping of habitats of conservation value in Norway. *Terje Klock, Directorate for Nature Management, Norway*

10.30-10.45 Coffee

10.45-11.10 Classification and mapping of habitat types in Iceland and evaluation of conservation value. *Sigurdur H. Magnusson, Icelandic Institute of Natural History, Iceland*

11.10-11.40 Mapping and nature evaluation in Estonia *Martin Zobel, Tartu University, Estonia*

11.40-12.00 Mapping of protected habitat types of the Nature Conservation Act in Finland. *Pilvi Pääkkönen, Finnish Environment Institute, Finland*

12.00-13.00 Lunch

Excursion

13.00-15.00 Grassland vegetation in the Danish landscape - challenges for monitoring and evaluation.

15.00-15.15 Coffee

15.15-15.40 Developing remote sensing for monitoring of selected habitats of the Habitats Directive. *Johan Abenius, Naturvårdsverket, Sweden*

15.40-16.15 Satellite monitoring of Estonian Landscapes. *Kiira Aaviksoo, Estonian Environment Information Center, Estonia.*

Thematic issues

16.15-17.00 Integration across scales and disciplines: Habitats, ecosystems and landscapes - ecotopes, biota and human interference. *Lars Eriksstad, NINA, Norway*

17.00-17.45 Classification and mapping: Raw attributes vs. classes. *Geoff Groom, DMU, Denmark*

18.00-19.30 Dinner

Workshop

19.30-21.00 Group discussions and summary on mapping and monitoring. Themes:

- Concepts and criteria for evaluation of conservation value.
- Habitats Directive: How do we map and monitor habitat types?
- Methods for mapping and monitoring: Data sources, cost-effectiveness.

Sunday November 3: “Monitoring and delivering”

Introduction

09.00 - 09.10 A request for reference-based monitoring - Can we make biological sense of the Habitats Directive and Water Frame Directive? *Flemming Skov, DMU, Denmark*

09.10 - 09.30 Natura 2000 monitoring – Can we satisfy national needs and EU demands at the same time? *Johan Abenius, Naturvårdsverket, Sweden*

09.30 - 10.20 What is the natural baseline for evaluation of condition and trends? - With special emphasis on the natural vegetation of North-western Europe *Jens-Christian Svenning, University of Aarhus, Denmark*

10.20-10.40 Coffee

Methods

10.40-11.20 How do we define favourable conservation status in forests and what are the relevant indicators for inclusion in monitoring programmes? *Jacob Heilmann-Clausen, KVL, Denmark*

11.20-12.00 Numerical methods for evaluation of biological condition in open land habitats *Rasmus Ejrnæs, DMU, Denmark*

12.00-13.00 Lunch

Thematic issues

13.00-14.00 A Dutch approach to the aggregation and processing of data for the achievement of national indicators – The Natural Capital Index. *Arjen Hinsberg, RIVM, Holland*

Closing session

14.00-15.00 Summary of monitoring and delivering. Recommendations and future work.

15.00 Departure

Participant list

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Classification and mapping of habitat types in Iceland and evaluation of their conservation values

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Introduction

Classification of habitat types has been ongoing in most European countries for some time, both within the European Council and the European Union. This work has been useful in nature conservation. In Iceland there has been an interest to follow this line of work in order to fulfil international and national nature conservation obligations. For the classification and description of Icelandic habitat types, we believe it is essential to use similar methods as in Europe to allow comparison within and between countries. Iceland is an isolated, volcanic island with species poor flora and fauna, which differ considerably from the neighbouring countries. A mainland habitat classification scheme can, therefore, not be adopted without revisions. Also, there is a general lack of information on several characteristics of Icelandic nature; the classification and definition of Icelandic habitat types calls for the broadening of existing data and intensive new field sampling.

In 1999, a special study, “Habitat project”, was initiated at the Icelandic Institute of Natural History. It is carried out in co-operation with the Master Plan for Hydro and Geothermal Energy Resources in Iceland. The main purpose of the project is to: define and describe habitat types in the highlands of Iceland; to determine their size and distribution; and to develop methods to access their conservation value. This work is intended to help in the evaluation of the conservation value of several proposed power project sites in the highlands of Iceland. The project is funded by the Energy fund, the National Power Company of Iceland, and the Icelandic Institute of Natural History.

Methods

The first phase of the work aims to classify habitat types in the central highlands of Iceland, an area of about 40.000 km². As the European methodology of habitat classification is based on vegetation, it was important to test if vegetation communities could be used as a base for an Icelandic habitat classification scheme. This was feasible because plant communities in two thirds of the country have already been mapped.

The project consists of four main parts: preparation, field studies, classification and description of habitat types, and conservation criteria and values.

Preparation of work

Based on the classification of European Union Habitats (The Interpretation Manual of European Union Habitats, 1996), the Palaearctic Habitat classification (Devilliers-Terschuren and Devilliers-Terschuren 1996, 2001), and information on Icelandic nature (vegetation, soil and animals), a list of hypothetical or preliminary habitat types was prepared for the central highlands of Iceland.

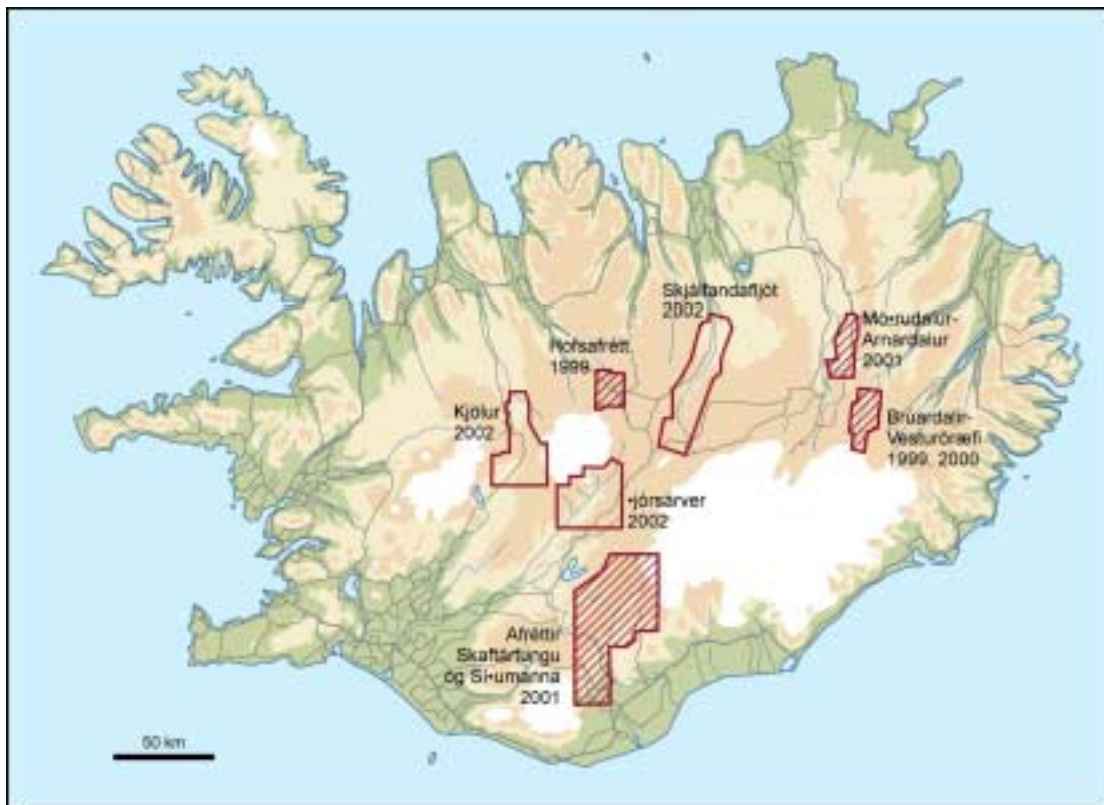


Fig. 1. Map showing the location of the study areas in the central highland of Iceland and the years of study. The present results are based on data from four of the areas (hatched).

Field studies

In order to test the working hypothesis, field studies were carried out in seven highland areas – measuring a total of 4000 km² – in 1999 – 2002 (Fig. 1). These areas were all chosen subjectively on the basis of two criteria. Firstly, the areas should include diverse vegetation types and secondly, each area should include one or more sites proposed for hydropower projects. Within each area similar methods were used.

In the field study a stratified random sampling was used. A vegetation map of each area was explored and the existing plant communities subjectively classified or transferred into preliminary habitat types. Based on this classification a new map of each area was produced showing the preliminary habitat types. Within the preliminary habitat types several points of study were randomly chosen.

In the field the randomized study points were located by using GPS instruments and vegetation transects, 200 x 2 m, were laid out. Along each transect 8 plots (1 x 0,33 m) were randomly chosen for sampling of vegetation and environment. In these plots the total cover of vegetation was estimated and also the cover of individual vascular plant species, the total cover of vascular plants, bryophytes, lichens, cryptogamic crust and exposed rocks. The above ground height of vegetation and the slope of land was measured and also the height a.s.l. In addition the topography was classified. The soil was classified to type and moisture and samples taken for measurement of pH and carbon. The soil depth was measured and the presence/absence of permafrost noted. Lichens and bryophytes were sampled along each transect for species identification. All additional vascular plant species found on the transects but outside the plots were also noted.

Land arthropods were collected on the transects in a few of the study areas by using pitfall traps and sweep netting.

Within each area density of breeding birds was studied along several km long line transects that crossed different preliminary habitat types.

Classification and description of habitat types

So far, data from four of the areas have been analysed and the results published in three different reports (Fig. 1) (Einarsson et al. 2000, Magnússon et al. 2001, 2002). The analysis includes a total of 260 vegetation transects and 460 km of bird transects.

Species and environmental gradients were analysed with Detrended Correspondence Analysis (DCA) (McCune and Mefford 1999) and the vegetation transects classified with two-way indicator species analysis program TWINSpan (Hill 1979). In both cases the analysis were based on the cover of vascular plants and the mosses *Racomitrium lanuginosum* and *R. ericoides*. The TWINSpan results were then used as a base for the classification of habitat types.

A short description of each habitat type was made based on the information on environmental factors, vegetation, soil, land arthropods and birds. The description consists of a name of the habitat type, general description, soil characteristics, characteristic plant and animal species, rare features, geographical distribution and a representative photograph.

A revised map of habitat types was drawn for each study area. A list of all the plant communities found within the area was inspected and the plant communities (dominant species, soil moisture classes, total plant cover) compared to the revised habitat types. Plant communities that were most similar to a particular habitat type were then grouped together to form a specific type. Based on these groupings, the map of habitat types was drawn.

Conservation criteria and values

Following the classification and mapping of habitat types, their conservation value was evaluated on the basis of 17 conservation criteria (Table 1). This evaluation was carried out by a group of experts. The habitat types were scored for each conservation criteria on a scale of 1 to 3. The sum of scores for each habitat type was then determined and used for comparison.

Table 1. Overview of the conservation criteria used to evaluate the conservation value of Icelandic habitat types.

Human centered criteria	Intrinsic values criteria	Ecological criteria
Economic value	Rarity	Diversity
Aesthetic value	Extremity	Wildness
Education value	Character value	Originality
Social/historical value		Continuity
Recreational value		Authenticity
Scientific value		Productivity
Spiritual/religious value		Disturbance tolerance

Results

Classification of the 260 vegetation transects from the four highland areas revealed 20 different habitat types. Within these types a great variation was found in vegetation cover and in several other characteristics. Six of these habitat types had very low vegetation cover. They were all confined to unstable surfaces like river flood plains, inland dunes, gravel flats and areas with volcanic ash. Two of the habitat types had intermediate cover values, one was restricted to snow beds but the other had high cover of *Racomitrium* mosses. All the other 12 habitat types had relatively high vegetation cover but varied greatly in soil and surface characteristics. They include habitat types on lava flows (1), dryland heaths (3), moist heathlands (2) and wetlands (6).

The species richness varied greatly between habitat types. The highest number of vascular plant species was found in dryland heaths habitat types, snowbeds and in palsa mires but the lowest on gravel flats and ash fields.

The species composition of land arthropods reflected the main pattern found in the

vegetation. Great difference was found between habitat types in terms of breeding density of birds. The total breeding density of birds was also positively related to vegetation production.

A large part of all the areas had habitat types with sparse vegetation cover. However, the relative size of many other habitat types differed greatly between areas probably reflecting difference in climate, type and density of bedrock, volcanic activity, erosion and livestock grazing. Habitat types rich in *Racomitrium* mosses were, for example, most common in the southern part of the highlands where the bedrock is permeable and precipitation high, while dryland heath habitat types were most common in the eastern highlands where the bedrock is relatively dense and precipitation low.

Conservation values of habitat types

As the study was limited in scope, conservation values of habitat types for the whole country can not be given yet. However, the present results indicate that palsa-mires, lava-flow-moss-heaths and alpine-heaths have the highest conservation values of the different habitat types in the highlands of Iceland.

Discussion

Classification of habitat types

The method used for analysis and classification of Icelandic habitat types has several advantages. It gives significant, new information about Icelandic nature and its characteristics. It provides valuable information on the distribution of several plant and animal species (vascular plants, bryophytes, lichens, land arthropods, birds) and the environment they are found in. Further, using standardized methods similar to those in use Europe is very valuable and enables direct comparisons.

The method has also several disadvantages. It is time consuming and therefore rather expensive. Because a limited number of vegetation transects can be sampled the most rare habitat types are probably overlooked. Another drawback of the method is how strongly it depends on the existing vegetation mapping data base. Errors in the vegetation mapping will be carried over into the habitat type map. Finally, as the classification is based on small patches of plant communities or units, the landscape level is probably somewhat overlooked.

Evaluation of conservation value

The experience of evaluating the conservation value of habitat types has its pros and cons. So far, the overall impression is that the method gives a broad overview of what is important to preserve or protect. However, the method is rather complicated, time consuming and needs broad knowledge. There are also limited guidelines to follow. It seems therefore urgent to improve the method. As the conservation criteria are numerous and overlap in several ways, a reduction of their number and clear definitions of their meaning should be of great use.

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Integration across disciplines and scales

Lars Erikstad & Vegar Bakkestuen

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Introduction

Mapping, monitoring and managing natural areas has a central part in modern planning concepts. An optimal nature management is dependent of relevant knowledge of the spatial occurrence of natural areas, species and ecological processes. As a part of the management, good monitoring systems has an important part in controlling the quality and results of the planning process relative to the actual changes in the landscape. It is likewise a crucial element in understanding and documenting environmental effects linked to man-made impacts as well as changes of natural systems, to help for better decision-making.

Ecological systems are spatially heterogeneous, exhibiting considerable complexity and variability in time and space. The development and maintenance of spatial and temporal pattern are crucial for the dynamics of populations and ecosystems. Environmental mapping, monitoring and management faces challenges linked to the understanding of complex systems and scale relations (e.g. Gustafson 1998; Cale & Hobbs 1994) and therefore contains major elements of interdisciplinary integration across scales. The choice of scale, overview or detailed, is difficult because methods and costs differ to a great extent and the linkage between regional data and ground truth in very detailed scales are often limited understood. Each individual and each species experiences the environment on a unique range of scales, and thus responds to variability individually. Thus, no description of the variability of the environment makes sense without reference to the particular range of scales that are relevant to the organisms or processes being examined (Levin 1992). Natural area mapping will never the less be of great importance to define spatial structures in the landscape and analysing natural recourses and habitat quality. This is the key to scaling and interdisciplinary integration: knowing what data and details is relevant on other scales, and what is noise.

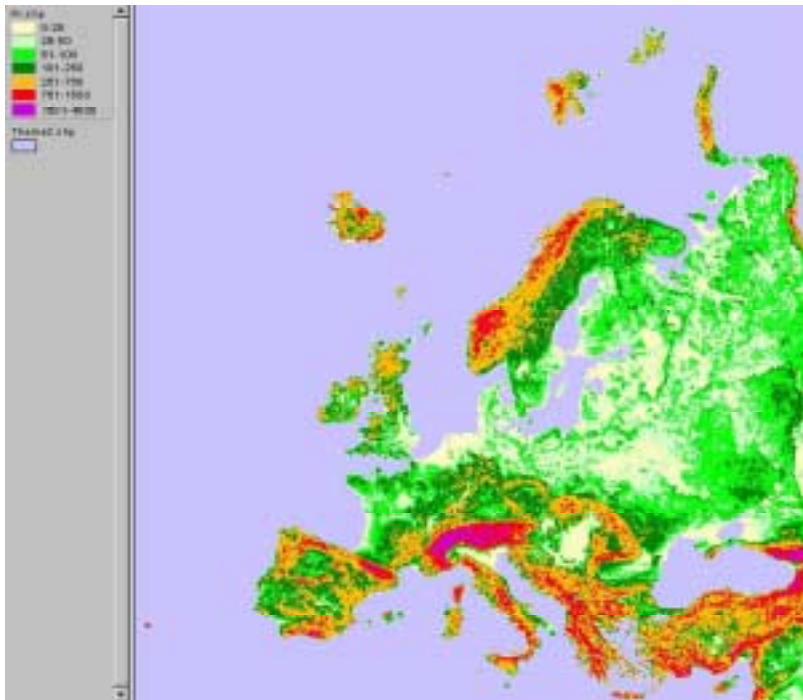


Figure 1. A digital elevation model for Europe.

Relevant data sources and analytic methods

Large databases consisting of existing geographical information (earth observation data, topographical data, digital land use data, cultural heritage data, vegetation maps etc) are

available for integration and comparison with other environmental data also from detailed scale levels. One of the most useful, but not often used data set available is digital elevation models (eg. Davis & Goetz 1990) which exists in different resolutions in most areas. Even the coarse resolution data sets which exist for most continents (1 km x 1km resolution, Hydro 1k Elevation Derivative Database, USGS) can yield ecological relevant information on a regional scale (**figure 1**). In combination to other data sets on the same scale such as land cover (CORINE), soil (UNESCO), forest cover (UNEP <http://www.grid.unep.ch/data/grid/gnv170.php>) and similar, increased understanding of the distribution of important landscape and ecological characteristics can be obtained (eg. Riitters et. al 1995; Iverson & Prasad 1998). This is useful for overall planning and monitoring on an EU level to a national and regional level, but may also serve as a background for more detailed studies as it provide a robust tool for stratification and classification.

Data also exist describing human activities. Mapped structures such as roads, railways, buildings and similar may also indicate the balance between human activity and natural processes. Such pressure indicators are in use indicating ecological state on a global scale (UNEP 2001). Used on a regional scale it is also possible to combine these data by different analytical techniques studying the spatial distribution of the human activities and the shape, interaction and quality of the land in between these structures (**figure 2**).

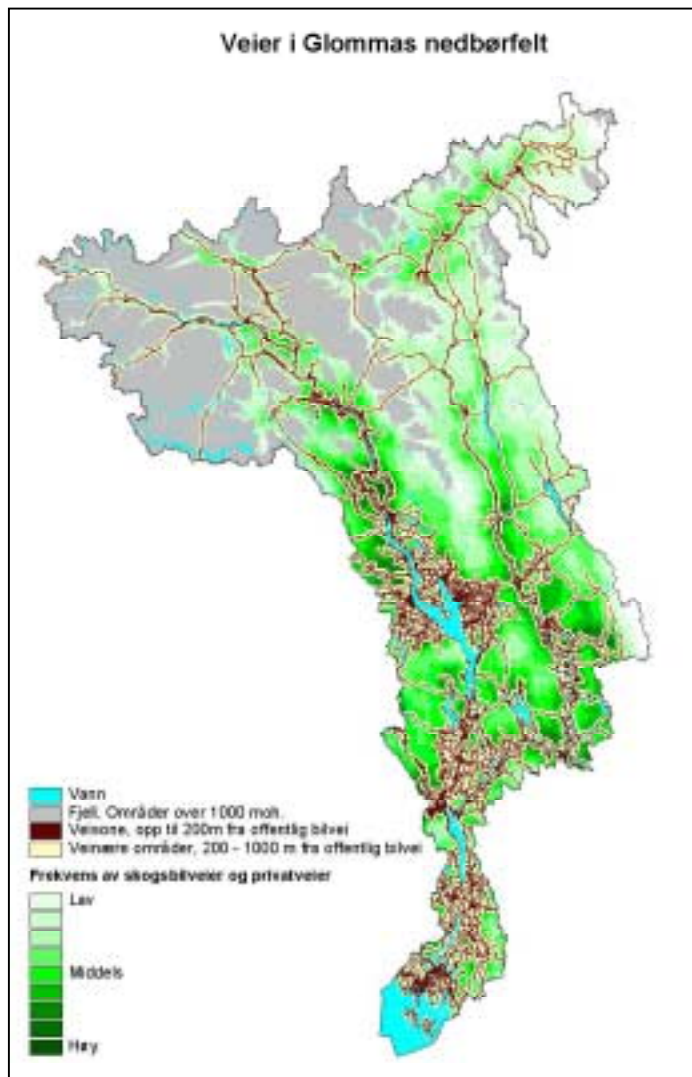


Figure 2. The position of major roads and density of smaller roads in the Glomma watercourse, south-east Norway.

Mapping of natural areas can be very resource demanding,

especially if these types are defined in great ecological detail. Utilising existing data linked to elevation, geology and vegetation will however provide possibilities for meaningful modelling efforts on a landscape level, which can improve considerably the mapping status

for large areas and serve as a low-cost improvement of knowledge status for land use planning and nature management (**figure 3**).

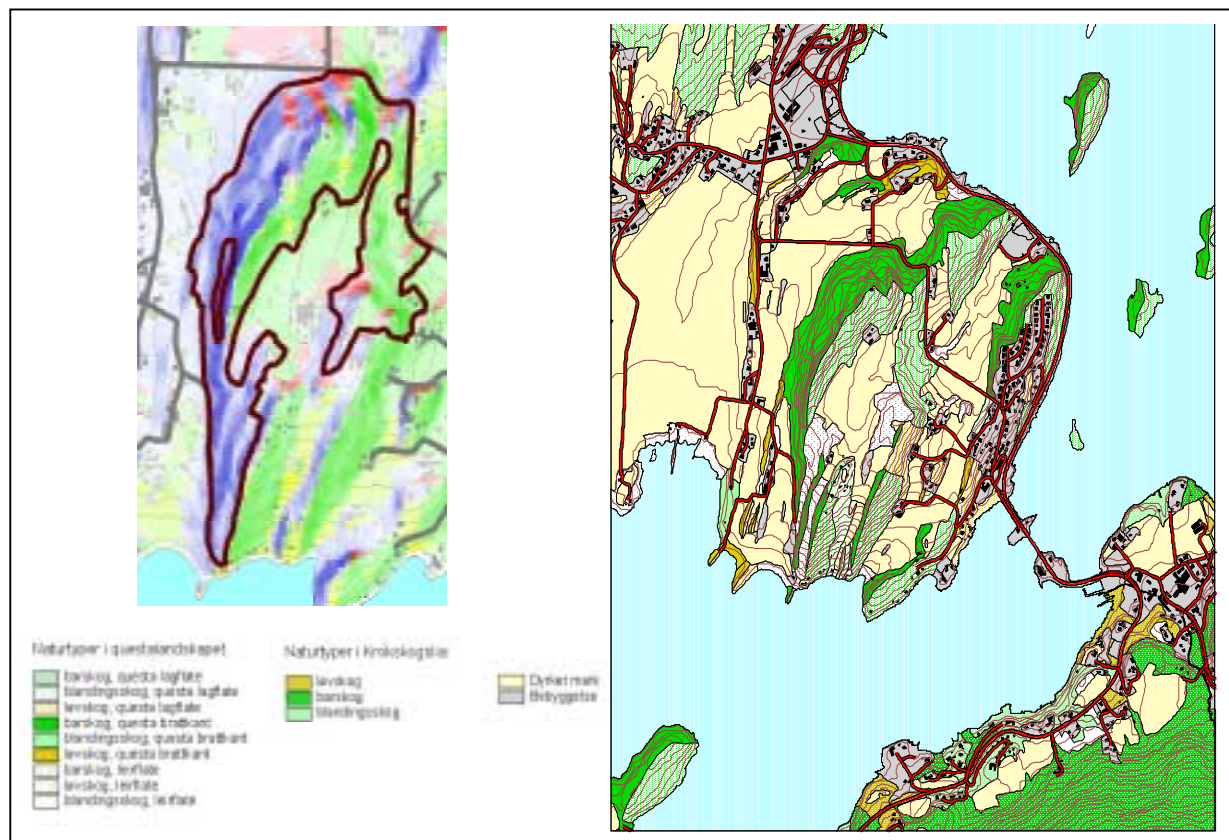


Figure 3. Mapping of natural resources as well as rural and urban activity in two landscapes.

Use of remote sensing has also an increasing role to play integrated in such activities. It will also serve as a useful and better starting point for detailed ecological mapping, inventories and monitoring. It is, however, important to stress that field validation is important for all such GIS-based mapping and modelling. It is also important to know what sort of uncertainty that is acceptable in different stages of the mapping. Normally, in grid GIS-based mapping, uncertainty is referred to as a grid cell uncertainty, that is the chance for a given information in a given point is correct or not. Dealing with natural and human made structures this kind of uncertainty may be rather great, but the result may still be useful. If the result describe well the structure and pattern with high accuracy, this may be as important as getting the right value cell by cell. Moreover such mapping techniques has the advantage to fast improve knowledge, but should be regarded as a step towards a goal, not a goal in itself.

Some times we are faced with a large scale impact factors that might cause changes in ecosystems on very different scale levels, also on detailed scale levels. This apply for long distance airborne pollutants and for climate change. In Norway, three long-term studies of species-environment relationships in boreal forests, with permanently marked plots, were thus initiated to study the effects on both a detailed and a regional scale level (Økland et. al 2001). 17 different areas distributed along main regional gradients over the whole country were selected to make up a national network of areas for intensive monitoring for vegetation changes in forests dominated by Norway spruce and birch (**figure 4**). Each area is investigated by a standardised methodology (Økland 1990, Økland 1996) and the vegetation analysis are conducted in 50 1m² sample plot and are reanalysed every fifth year. The results from the detailed scale level reveal two patterns of changes in biodiversity that could

be related to broad-scale impacts. In spruce forests on richer soils in the southern part of the country, the abundance of several vascular plant species has declined due to long transported air pollution. The abundance of most bryophyte species has increased over most of Norway in the 1990s due to climatic conditions particularly favourable for bryophyte growth (Økland et al. 2001).

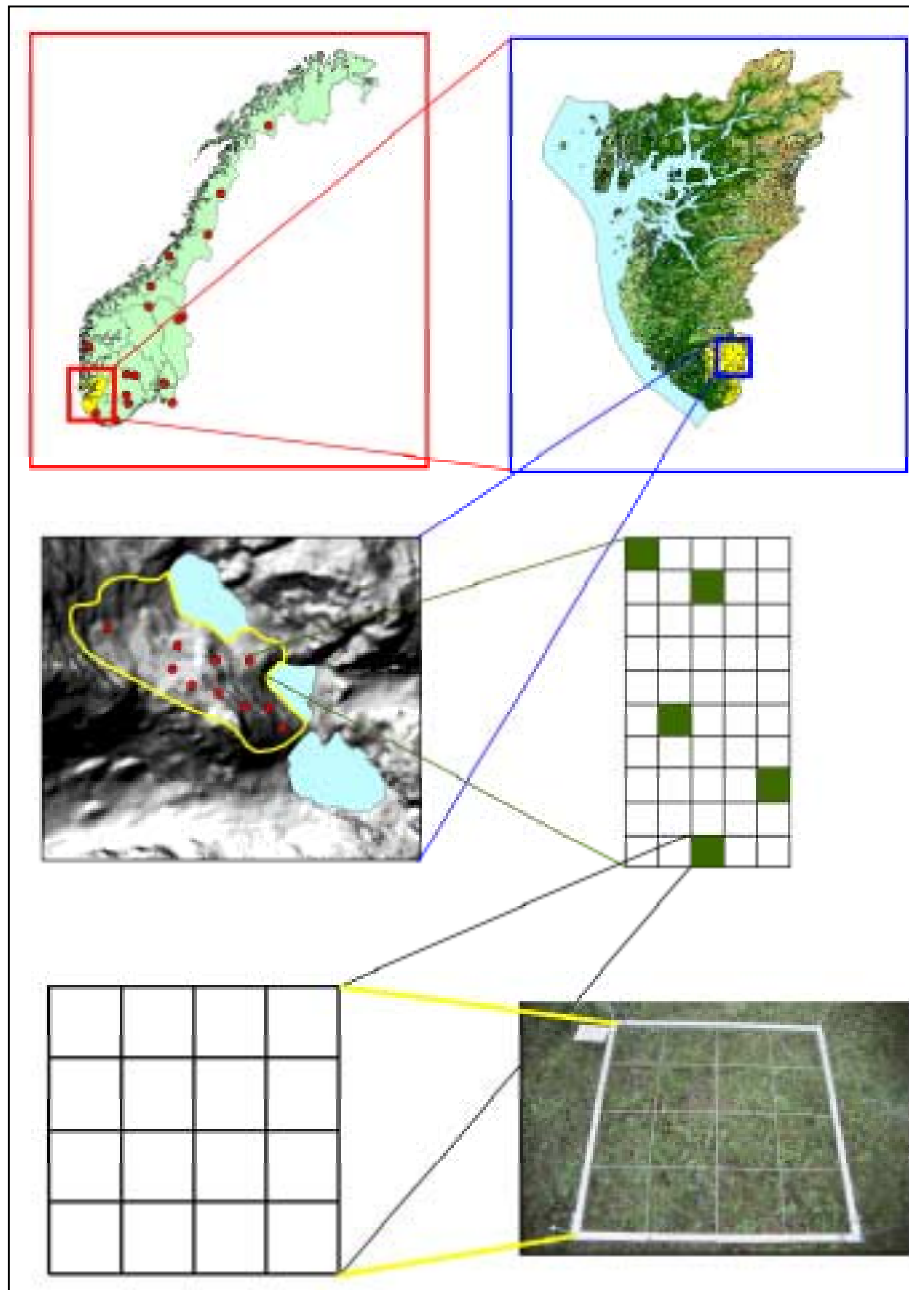
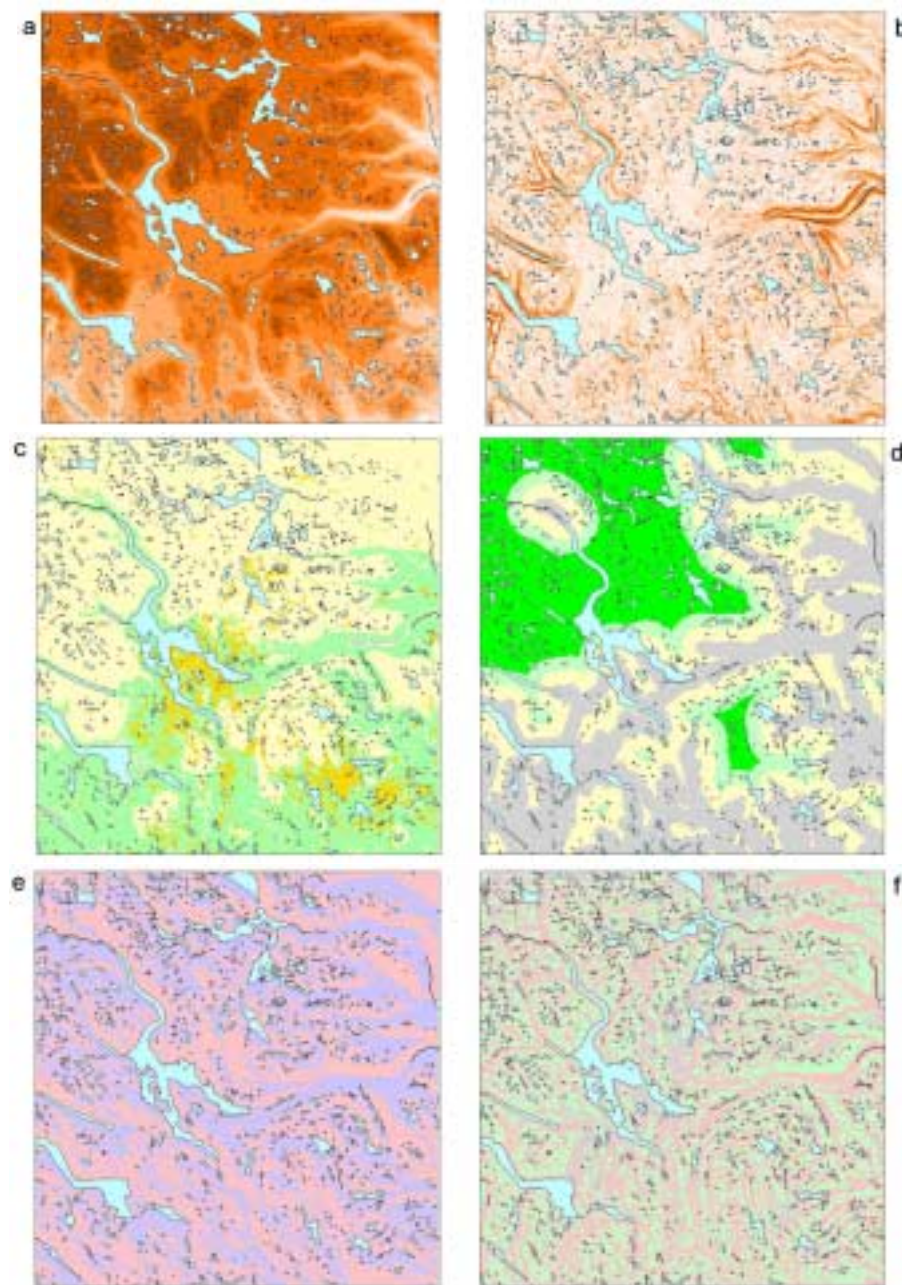


Figure 4. *The linkage between detailed inventories of sample plots and large-scale patterns.*

These results demonstrate that the concept used for intensive monitoring of forests in Norway enables early

detection of changes in vegetation brought about by broad-scale, regional, impact factors. Another upscaling example is described by Munier et al. 2001 based on the same type of sample plot data. The upscaling was based on natural areas and gave predictions of semi-natural grasslands with an accuracy of up to 87%.



Using a hierarchical field design (eg. Dooley & Bowers 1998) and/or having same type of data on the different scale levels (for example raster maps, aerial photographs, satellite data and digital terrain model with different grain sizes), will contribute to “controlled” up- and downscaling (for example produce correct estimates of the effects of changes in the landscape, and improve relevant indicators for environmental change based on aerial and satellite images). Increased knowledge of such scale relations will be of great importance when data from these monitoring schemes shall be interpreted and extrapolated to indicate changes for entire landscapes.

Figure 5.

We have performed an analysis relevant for up- and downscaling of data by integrating data from different sources: (a) the Norwegian 100m x 100m digital elevation database, (b) topographical maps in the scale 1:250 000 and (c) derivatives from these sources (**figure 5**). This analysis was performed in Møsvatn, which is one of the 17 areas in national network of areas for intensive monitoring for vegetation changes in forests (Økland 2001). The data was analysed by a PCA by summarising the variables (derivatives) in grids (grain) of different sizes (**figure 6,7,8,9**) to identify the major gradients existing in the material in the different scales. PCA techniques has been used in GIS studies in ecosystem classification by quantifying and integrating environmental variables (Host et al. 1996, Baker & Weisberg 1995). The importance of regional gradients (elevation/temperature) and local scale gradients (nutrient state, local hydrology, topography) was related to the spatial resolution of the sampling.

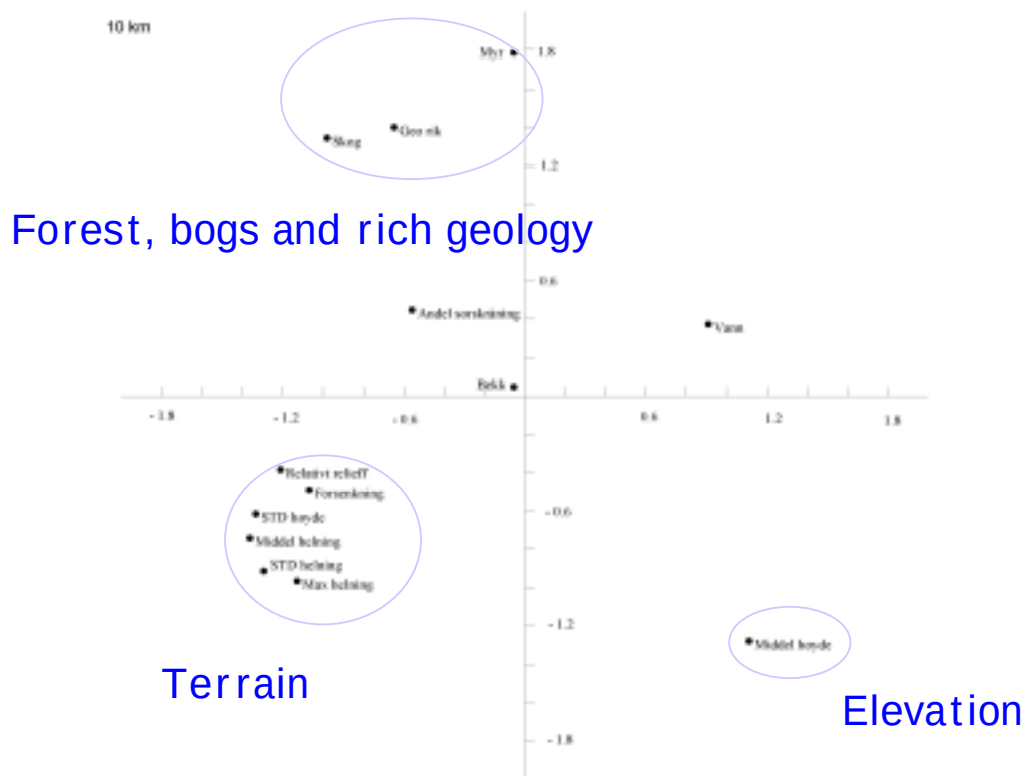


Figure 6. PCA-analysis of 10 km squares

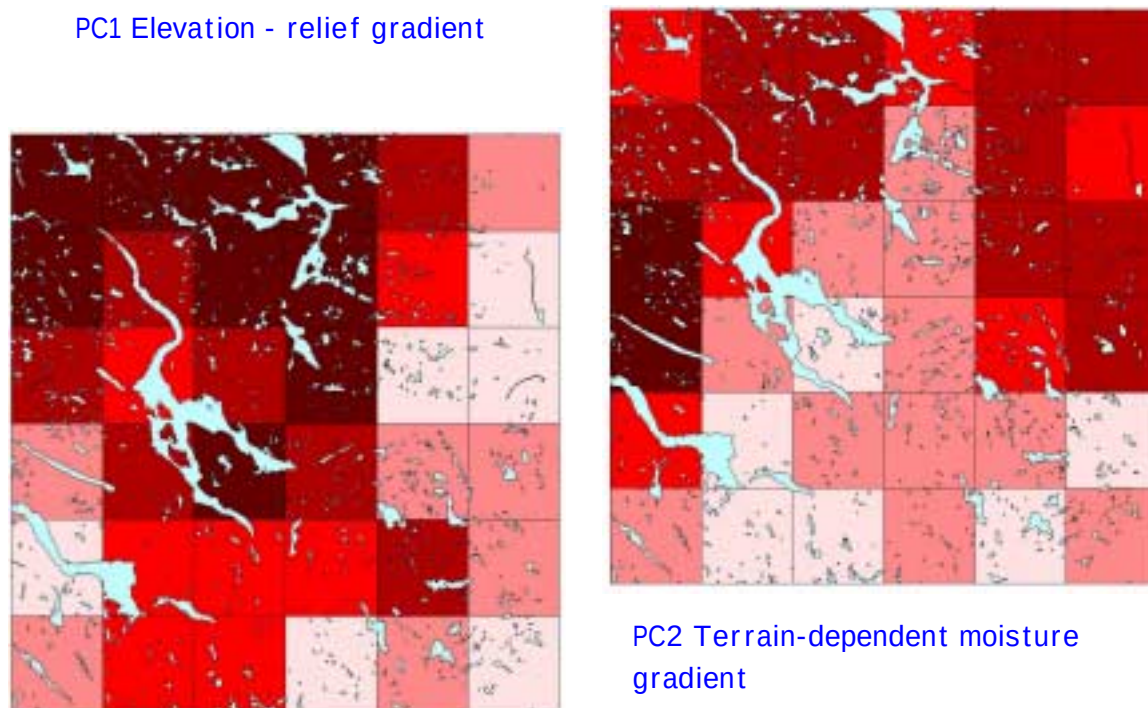


Figure 7. Major variation in 10 km-squares

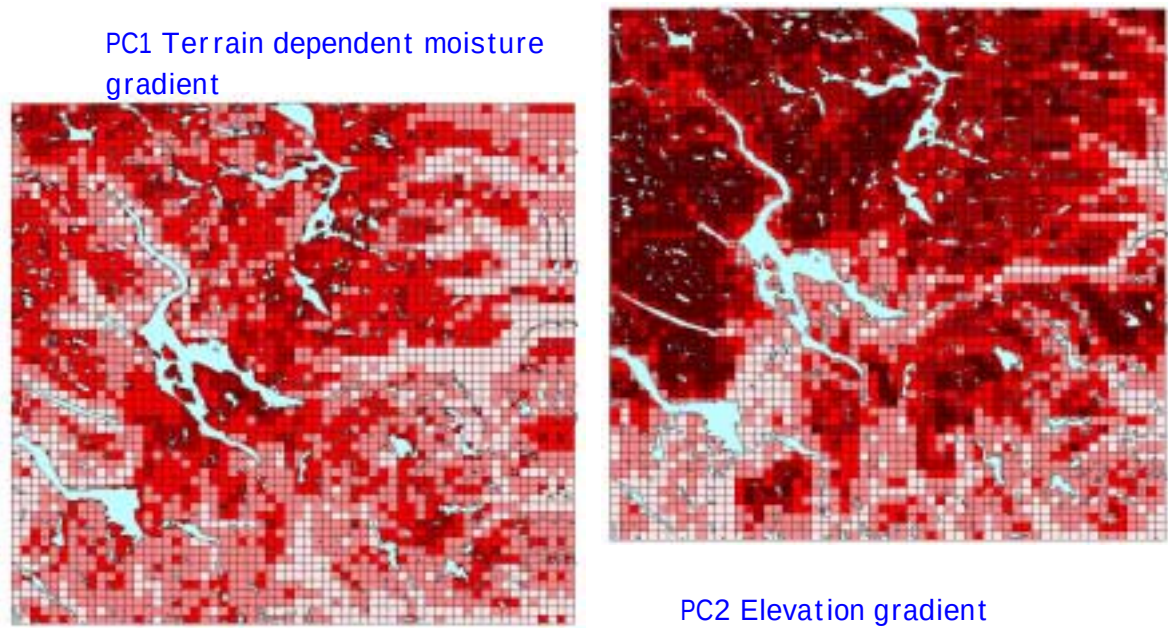


Figure 8. Major variation in 1 km-squares

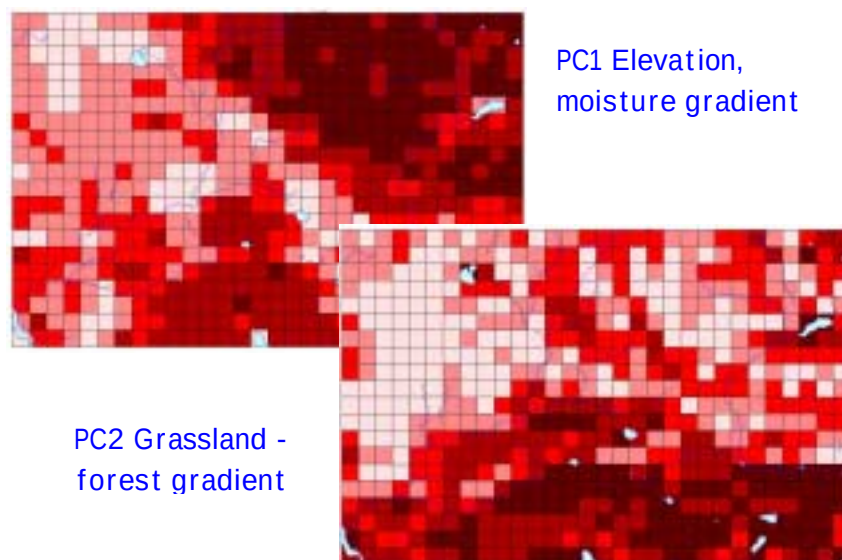


Figure 9. Major variation in 100 m-squares

Typically, regional gradients dominate at a spatial resolution (grain) down to approximately 5 km, and local gradients up to resolutions of 0.5 – 1km. In the course scale (grain sizes of 10 – 20 km), elevation in itself was most important, while in medium scales (1 – 5 km) other terrain indices like slope and relative relief dominated. In the detailed scales (0.1 – 0.5 km) local topographical conditions linked to hydrological conditions was highlighted. It is interesting to see that elevation and terrain data seems important on all scales although linked to different ecological attributes. This relationship is important for both the possibilities to link data from different scales and different disciplines. Another important relationship was discovered when relating vegetation composition in the sample plot to terrain parameters derived from 10 m raster model. We found a significant relationship between the elevation, slope and aspects from the GIS model and the species composition in the sample plots (Bakkestuen & Erikstad 2002). Such relationships are also important if results from

the detailed scale level (sample plots) should be extrapolated in prediction models valid for larger areas.

Data aggregation (up-scaling) can affect the results significantly (Bian 1997) and therefore also predictions drawn on the background of these. If spatial autocorrelation is present before data aggregation, this will often increase R²-values in later regression analysis (Walsh et al. 1994). Downscaling of information is also closely tied to the spatial statistical properties of the data sets and results have to be interpreted with this problem in mind. The growth in technologies relevant to spatial analysis has given a great potential to extend ecological modelling from local, intensive investigations to regional/global systems. In connection with validation of global products based on earth observation it is established procedures for comparing and utilising data from different scales at the landscape level (Cohen and Justice 1999, Thomlinson et al 1999).

Discussion and conclusion

Discussing the matter of scale, the size of the investigation area can be defined as "extent", whereas "grain" is defined as the size of individual sample units (Wiens 1989; Gustafson 1998). We can not generalise outside the extent without accepting an assumption of scale independence between patches and processes. We neither can discover objects smaller than the grain. We are often forced to increase grain if we shall increase the size of the investigation area. This implies that, if we try to capture macro-scale patches, it will be at the expense of micro-scale variation. When the scale of measurement of variables changes, the internal variance in this variable will also change. A constant extent and an increase in grain will lower the spatial variance. A constant grain and an increase in extent will raise the variance. This implies that the question of up- and downscaling represent a major challenge in understanding complex systems and scale (Quattrochi & Goodchild 1997). Recent studies presented by Pereira (2002) suggest a classification system of different temporal and spatial scale relations.

Landscape ecology is based on the premise that there are strong links between ecological pattern and ecological function and process (Hersperger 1991). To understand how the landscape is build up and how it changes, it is important to analyse its composition, structure and function in terms of different disciplines (geology, geomorphology and biology). Composition is described as occurrence and proportions of nature areas (habitat types) and patterns of dispersing species. Structure is given by physical patterns of the elements (components) of the landscape measured as heterogeneity, connections, dissimilarity, distribution of frequency, etc. Functions include ecological and evolutionary processes such as disturbance, habitat change, energy fluxes, nutrition cycles, erosion, hydrological processes and changes in the human use of the landscape (Franklin 1988; Noss 1990; Gustafson 1998). Noss (1990) presents a hierarchical scale model for (bio-)diversity at four different scale levels: regional landscape level, community-ecosystem level, population/species level and genetically level.

Composition, structure and function of the landscape at a regional scale level affect and get influenced by other levels in a scale model. In monitoring of biodiversity (e.g., the Gap Analysis Program [GAP]), input data often come from vegetative features and categories of land use mapped at coarse spatial scales. This implicitly assumes that species richness data collected at the coarse scales provide first-order approximation information on the more detailed scale levels. Conroy & Noon (1996) list some problems associated with this assumptions (1) the species abundance distributions and species richness are poor surrogates for community/ecosystem processes, and are scale dependent; (2) species abundance and richness data are unreliable because of unequal and unknown sampling probabilities and species-habitat models of doubtful reliability; (3) mapped species richness data may be inherently resistant to "scaling up" or "scaling down"; and (4) decision-making based on mapped species richness patterns may be sensitive to errors from unreliable data and models, resulting in sub-optimal conservation decisions. Conroy & Noon (1996) suggests an

approach in which mapped data are linked to management via demographic models, multiscale sampling, and decision theory. On the other hand, a scale-independent measure of species abundance is developed by Kunin (1998) by using presence-absence maps at varying spatial resolutions. By extrapolating "scale-area" curves, Kunin (1998) & Kunin et. al (2000) show that species abundance can be estimated accurately even at scales finer than those used to parameterise the model. This is an important step forward in linking the landscape level with detailed scale levels.

The complexity and variability of landscape structure is typically represented by categorical maps or by a collection of samples taken at specific spatial locations (point data). Categorical maps quantize variability by identifying patches that are relatively homogeneous and that exhibit a relatively abrupt transition to adjacent areas. Alternatively, point-data analysis (geostatistics) assumes that the system property is spatially continuous, making fewer assumptions about the nature of spatial structure. Gustafson (1998) reviews the two techniques and conclude that pattern analysis techniques (categorical maps) are most useful when applied and interpreted in the context of the organism(s) and ecological processes of interest in appropriate scales (which as he points out can be unknown). Point-data analysis, however, can answer two of the most critical questions in spatial pattern analysis: what is the appropriate scale to conduct the analysis, and what is the nature of the spatial structure? These properties brings novel tools to ecology for the interpretation of spatial patterns of organisms, of the numerous environmental components with which they interact, and of the joint spatial dependence between organisms and their environment (Rossi et al. 1992). Geostatistics and other statistical methods are in this way important in order to utilise the whole potential in digital spatial data analysis. Data are often correlated in space (and time); spatial structures can emerge from different sources such as measurement errors, continuity effects including spatial heterogeneity, and spatially dependent processes and mechanisms (Haining 1990). In future work it is important to design mapping and monitoring systems and the choice of indicators in a way that makes the data suitable for geostatistical analysis.

Natural area mapping on coarse scales is important for regional planning, but also for detailed planning and research as controlled reference as well as forming the best possible basis for stratification in scientific detailed studies. In natural area mapping, multidisciplinary integration of data as well as scale studies, terrain data are central both because of their availability and their ecological relevance. It should, however, be emphasised that care should be taken in basing species management based on modelled up- or downscales data where the process of scaling is not controlled by relevant environmental data from both (or all) used scales.

An improved understanding of how knowledge based on data from one spatial scale is connected with knowledge based on data from another scales is important for integration of data across scales and disciplines. This is fundamental for utilising spatial data in monitoring programs and in the process of acquiring information from ground truthing. This is also fundamental for natural resource management in a realistic spatial context.

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Classification and mapping: Raw attributes vs. classes.

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Abstract

Programmes for the mapping and monitoring of natural areas generally encounter, sooner or later, the following methodological questions: “what classes shall we use to map / monitor our natural areas?” and “how do the classes used to map / monitor natural areas in country (or study) X relate to those used in country (or study) Y?”. This paper explores, in a generic context, some of the issues involved in answering the above questions. It focuses upon the solutions provided through the implementation of a standardised, classifier-based classification system for legend generation. This is illustrated by the Land Cover Classification System (LCCS) that has been developed and adopted by the FAO/UNEP Africover project. Results are presented from use of the LCCS for analysis and comparison of five Nordic land cover related class sets.

Introduction

Classification, or “the ordering or arrangement of objects into groups or sets on the basis of their relationships” (Sokal, 1974), is a fundamental scientific operation for the simplification of complex <realities>. The aim of classification is to structure knowledge in order to facilitate communication and exchange. Classification is required for data recording, analysis and reporting. As such, classification is a <fundamental> task when faced with a <reality> as complex as a natural ecosystem and a need to map or monitor that <reality> in a broad but also detailed way. The success of the classification that is applied can massively influence the usefulness of the mapping and monitoring results. Class sets for mapping and monitoring of natural areas take many forms and cover many aspects of the complex, e.g. its vegetation, habitats, land cover and/or nature quality. For each of these various possible class sets are already available, such as the Corine Land Cover classes (CEC 1993), the EUNIS habitat classification (Davies and Moss 1997), the EU Habitat Directive’s Annex-1 habitat classification. Natural area mapping and monitoring projects will be required to justify the set of classes they use. Furthermore, increasing demands for international data, such as needed by the EEA, require harmonisation of divergent class sets. Thus, natural area mapping and monitoring programmes are often faced by the following questions:

“what classes shall we use to map / monitor our natural areas?” and

“how do the classes used to map / monitor natural areas in country (or study) X relate to those used in country (or study) Y?”

This paper builds upon work undertaken in 2000-2002 as part of the Nordic Council of Ministers NMD working group project “Development of remote sensing for Nordic terrestrial landscape monitoring” (<http://nordlam.dmu.dk>). The issue of class sets is fundamental to mapping and monitoring work using remotely sensed data, since raw image data are often simplified for subsequent application in the form of thematic classes, such as land cover. However, the issues discussed in this paper are generic, with relevance beyond just those with a remote sensing focus. Moreover, as noted above, “classification” is a core scientific activity that sits across specific areas such as ecology or environmental science.

part 1: the classification system



'SCAPE SWEETS' MAKES SMALL SWEETS - MANY DIFFERENT SORTS :



IN THE PAST IT HAS ALWAYS SOLD ITS SWEETS IN MIXED PRE-PACKED BAGS

BUT NOW THE MARKETING DEPARTMENT HAS DONE SOME RESEARCH THEY FOUND THAT THE COMPANY COULD INCREASE PROFITS BY SELLING ITS SWEETS, NOT IN MIXED PRE-PACKED BAGS, BUT SINGULARLY, WITH EACH SWEET PRICED INDIVIDUALLY. THE SALES DEPARTMENT HAS DECIDED THAT EACH SHOP SHOULD SET ITS OWN PRICES PER SWEET - BUT THIS SHOULD FOLLOW CERTAIN GUIDELINES.

THE MARKET RESEARCH SHOWED THAT THE PRICE OF A SWEET SHOULD BE BASED UPON :

THE COLOUR



THE LENGTH OF THE MAJOR AXIS



WHETHER OR NOT THERE IS A NUT INSIDE



THUS, THERE ARE THREE CLASSIFIERS, WITH THE FOLLOWING ASSOCIATED SETS OF STATES :

COLOUR

(a Categorical classifier)

- RED
- GREEN
- YELLOW
- BROWN
- WHITE

LENGTH OF MAJOR AXIS

(a Cartesian classifier)

- LOW : LESS THAN 3 MM
- MEDIUM : 3 - 7 MM
- HIGH : GREATER THAN 7 MM

NUT INSIDE?

(a Boolean classifier)

- NUT
- NO NUT

THIS IS THE **CLASSIFICATION SYSTEM** (non hierarchical)

part 2 : the nomenclature

THIS CLASSIFICATION SYSTEM IS ASSOCIATED WITH THE FOLLOWING **NOMENCLATURE** (set of all possible classes) :

red, short, nut	green, short, nut	yellow, short, nut	brown, short nut	white, short nut
red, short, no nut	green, short, no nut	yellow, short, no nut	brown, short, no nut	white, short, no nut
red, medium, nut	green, medium, nut	yellow, medium, nut	brown, medium, nut	white, medium, nut
red, medium, no nut	green, medium, no nut	yellow, medium, no nut	brown, medium, no nut	white, medium, no nut
red, long, nut	green, long, nut	yellow, long, nut	brown, long, nut	white, long, nut
red, long, no nut	green, long, no nut	yellow, long, no nut	brown, long, no nut	white, long, no nut

part 3 : the legend

TO SET ITS PRICES PER SWEET, EACH SHOP HAS TO APPLY THE CLASSIFICATION SYSTEM AS BEST IT CAN

STAN'S MARKET SWEET STALL :

can use vision to apply the COLOUR classifier
can use a ruler to apply the LENGTH classifier
but has no way of applying the NUT classifier (without also destroying the sweet!)

Stan's **LEGEND** is:

red, short	green, short	yellow, short	brown, short	white, short
red, medium	green, medium	yellow, medium	brown, medium	white, medium
red, long	green, long	yellow, long	brown, long	white, long

BILL'S SNAZZY SWEET EMPORIUM :

can use vision to apply the COLOUR classifier
can use a ruler to apply the LENGTH classifier
has an X-ray scanner... and can therefore also apply the NUT classifier

Bill's **LEGEND** is :

red, short, nut	green, short, nut	yellow, short, nut	brown, short, nut	white, short, nut
red, short, no nut	green, short, no nut	yellow, short, no nut	brown, short, no nut	white, short, no nut
red, medium, nut	green, medium, nut	yellow, medium, nut	brown, med., nut	white, med., nut
red, med., no nut	green, med., no nut	yellow, med., no nut	brown, med., no nut	white, med., no nut
red, long, nut	green, long, nut	yellow, long, nut	brown, long, nut	white, long, nut
red, long, no nut	green, long, no nut	yellow, long, no nut	brown, long, no nut	white, long, no nut

Figure 1. An illustration of the distinction between 'classification system', 'nomenclature' and 'legend'.

Classification

Class sets can be derived *a posteriori* by similarity and dissimilarity analysis of raw data, such as species lists, as is the case for phytosociological vegetation classification after the Braun-Blanquet method. However, whilst adaptable to the data set and the reality that it represents, *a posteriori* classification is unable to define standardised classes (Di Gregorio and Jansen, 2000). Class sets are more often defined *a priori*, with the greater potential for standardisation but also the possibilities that some objects cannot be easily assigned to a single class. This paper addresses issues concerning just *a priori* classification and class sets.

A widely used concept of *a priori* classification has four sequential stages (Eurostat, 2001):

- demarcation of the domain of the classification (the ‘universe of discourse’) – this is generally discipline and interest orientated
- establishment of a classification system of all objects in the universe of discourse
- establishment of a ‘nomenclature’ for coding and describing classes derived from the classification system
- defining of a ‘legend’ that is a subset of the nomenclature for use for a specific purpose with a specific set of data and techniques (such as mapping at scale 1:100,000 by automatic classification of satellite images); this stage implies also the setting-up of procedures for assigning objects to legend classes, i.e. ‘identification’

The use of these three terms is illustrated in Figure 1.

Sound classification is associated with the following characteristics:

- **Completeness:** The classification system exhaustively describes that segment of reality that is its domain. This also implies that the system is spatially consistent in its operation, i.e. that it is not designed from the perspective of just one or other region of the domain.
- **Absence of overlap:** The nomenclature classes are mutually exhaustive, without overlap. This implies that legends should not include ‘mixed’ classes; however within geographic contexts a distinction needs to be noted between such semantically or thematically mixed classes and the possibility for cartographically (spatially) mixed classes, whereby two distinct thematic classes can co-occur within a mapping unit.
- **Use of an underlying common concept throughout the classification system.** For example, use of a physiognomic-structural concept for land cover classification, a life-form concept for vegetation classification, or a functional concept for land use classification.
- **Use of a standardised classification system, comprising inherent classifiers that are clearly defined in terms of their states, consistently applied at the same hierarchical levels, and linked by a clear set of rules.** ‘Inherence’ means that only those characteristics that can be used to describe the domain objects in relation to the underlying concept and unambiguously are used. Thus ‘climate’ cannot be considered as an inherent classifier for either a life-form concept of vegetation or a physiognomic-structural concept of land cover. Whilst non-inherent characteristics may not be placed in the system as core classifiers, they may be included as additional descriptive attribute.
- **Independence in the classification system from data collection and processing tools:** The classification system should not be associated with a specific mapping or monitoring methods or scales, to ensure that as mapping tools develop the classification is able to adapt too.
- **Temporal consistency:** The classification system should relate to the state at the instant of observation, and not past or future states. (This does not prohibit the consideration of ‘change’ as the underlying principle for a specific classification system.)

Thus, there is much more involved in making a classification than simply choosing a set of desired or useful classes. Unfortunately however, that has been how many mapping and monitoring have operated, albeit with consideration of general ideas of how the objects involved should be arranged and drawing upon the strengths and weaknesses of related class sets as their starting point. The resulting class sets may be highly structured (i.e. hierarchical) but lack many of the principles and characteristics of a sound system. The IGBP-DISCover, the CORINE Land Cover and the UN-ECE land cover / land use 'nomenclatures' are such cases (Eurostat 2001, Di Gregorio and Jansen 2000). For example, the common confusion within class sets between land use and land cover objects, such as occurs in CORINE Land Cover, can be seen as a failure to apply a clear underlying common concept.

As well as providing a good basis for a new class set, the principles described above also provide a basis for combining and comparing data from different existing mapping and monitoring operations that have used different class sets. The intellectual and financial investments that have been made in existing data sets that use different class sets make it unrealistic to think in terms of 'discarding the old' on account of its inconsistency (Wyatt & Gerard, 2001). Moreover such archive data is immensely valuable for studying long-term change patterns. Furthermore, it is unrealistic to think in terms of using now and into the future a single standardised class set, since in accordance with differences in the prevalent universe of discourse, different classification of the same domain will continue to be required. Thus, class set harmonisation (or 'correspondence') is more relevant than class set standardisation. A standardised reference classification system, established in accordance with the above principles, provides the basis for operation as a bridging tool between class sets.

Certain areas of natural geoscience have successfully established strong classification. Since at least the 1970s soil science has adopted and developed such principles for establishment of a standardised reference system, resulting today in general harmonisation of soil classes across the world and increased understanding between soil scientists (U.S. Soil Conservation Service, 1975; FAO/UNESCO, 1988; FAO, 1998).

'Closer to home' for the mapping and monitoring of natural areas, the FAO/UNEP Africover project has during the late 1990s developed and adopted the Land Cover Classification System (LCCS), which represents a standardised classifier-based reference system for global land cover classes (Di Gregorio and Jansen, 2000).

The Land Cover Classification System (LCCS)

The LCCS (<http://www.lccs-info.org>) comprises a hierarchical rule-based set of classifiers and associated attributes to describe land cover in terms of its physiognomy and structure. This concept distinguishes between pure classifiers (e.g. life form, surface aspect, height, leaf type) that relate to inherent characteristics of land cover, and non-inherent environmental (e.g. altitude, land form, soil type) and technical (e.g. crop type, built-up object type, floristic aspect, salinity) attributes. As such, the LCCS provides a basis for class description, legend generation and also analysis of similarity and differences between legends (the so-called Translator Module), ie. class harmonisation.

In addition to the Africover project, LCCS has been applied for national land cover mapping for Bulgaria (Travaglia et al. 2001), national land cover mapping for Romania (FAO project, in progress) and Albania (Agrotec S.p.A. Consortium, in progress), the Global Land Cover 2000 project (<http://www.gvm.sai.jrc.it/glc2000/defaultGLC2000.htm>), and, as described below, the NMD NordLaM project. As a 'classification system' in the sense discussed above, the LCCS does not simply present to the user a specific set of classes, but instead it presents the potential for defining several thousand classes, since classes are determined as an outcome of using the LCCS classifiers.

In LCCS, an initial dichotomous phase uses three boolean classifiers: presence or absence of vegetation, edaphic condition, and artificiality of the cover. This results in the following eight major categories:

- A11 cultivated and managed terrestrial areas
- A12 (semi) natural terrestrial vegetation
- A23 cultivated aquatic or regularly flooded areas
- A24 (semi) natural aquatic or regularly flooded vegetation
- B15 artificial and associated surfaces
- B16 bare areas
- B27 artificial water bodies, snow & ice
- B28 natural water bodies, snow & ice

Further classifiers are then used for each of these 8 major categories in a modular-hierarchical manner. This second phase is customized to each of the eight major categories (Figure 2). Figure 2 shows the distinctions made in LCCS between pure (i.e. inherent) land cover classifiers (in blue), environmental attributes (in purple) and technical attributes (in green)

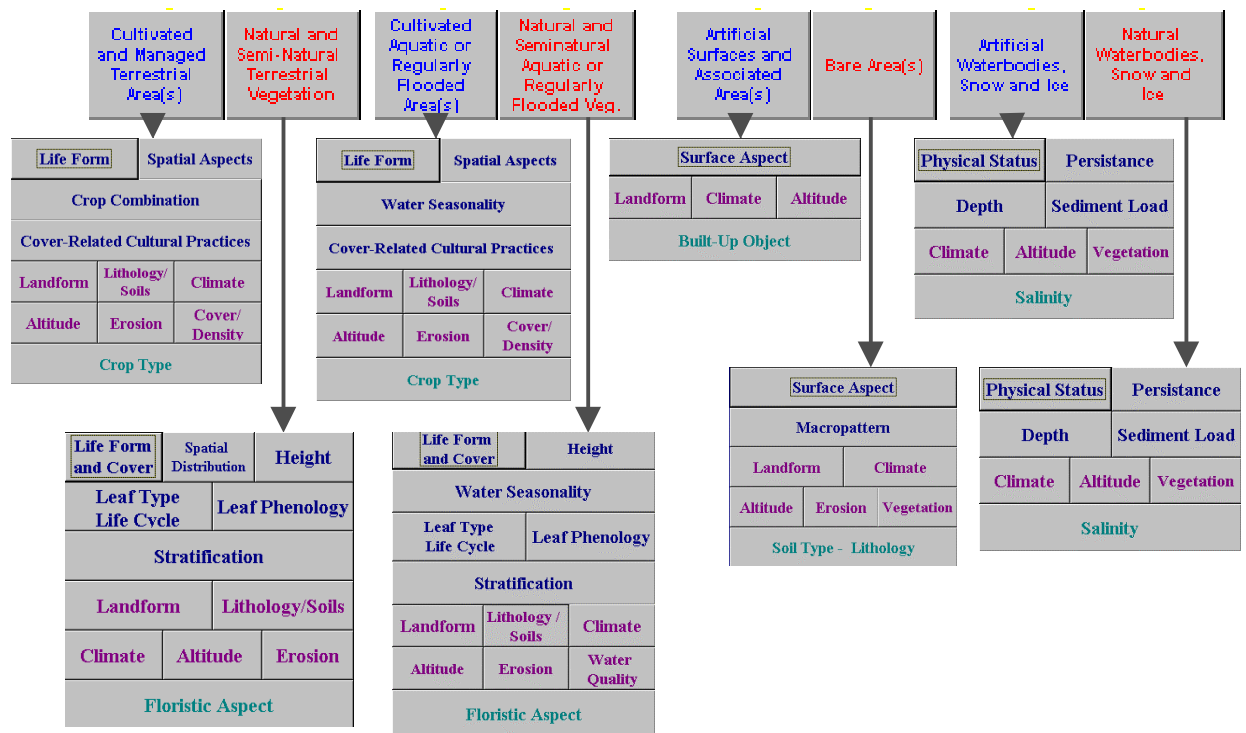


Figure 2. The structure of classifiers and attributes used in the modular hierarchical phase of the LCCS for the eight major land cover categories. (after Di Gregorio and Jansen, 2000).

An example of the modular-hierarchical phase for the major category B12 is shown in Figure 3. Use of the classification is rule-based, so, for example, in Figure 3, it is not possible to describe an object in terms of its ‘water seasonality’ without having also already described it in terms of its ‘life form and cover’ and its ‘height’.

Thus, LCCS is a new language to describe in a standardised way the land cover features. In a language, words and a syntax allow creation of a semantic concept. The different combination of words used with a given syntax give a wide possibility of concepts’ generation. In the LCCS, the *classifiers* are the *words*, the *classification rules* are the *syntax*

and the *land cover features* are the *concepts* to be described. as noted above, no pre-defined class list is provided by LCCS. Once described within LCCS land cover features are expressed by LCCS in terms of their boolean formula, a standard class name and a code (Table 1). The LCCS, with its associated software provides a basis for:

- class description (LCCS classification module)
- legend production and output (LCCS legend module)
- legend translation, i.e. class correspondence (LCCS Translator Module)

Figure 3. The LCCS modular-hierarchical phase for the major category B24. (The right hand frame shows the sets of states associated with just the first three classifiers.)

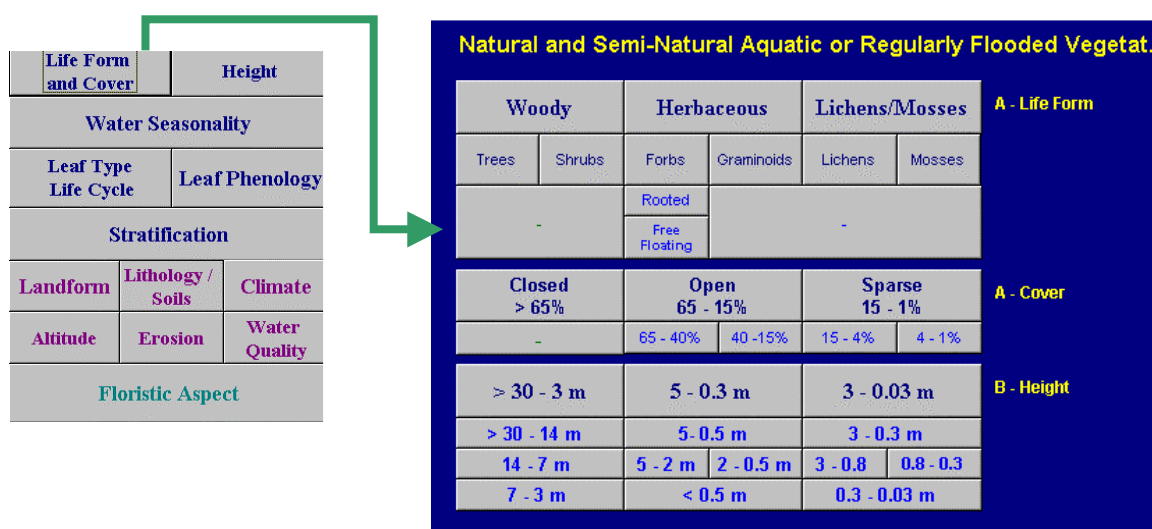


Table 1. The LCCS class feature output data.

Example: "Natural and semi-natural terrestrial vegetation (A12)"			
Classifiers Used:	Boolean Formula:	Standard Class Name:	Code:
Life Form & Cover	A3A10	Closed Forest	20005
Height	A3A10B2	High Closed Forest	20006
Spatial Distribution	A3A10B2C1	Continuous Closed Forest	20007
Leaf Type	A3A10B2C1D1	Broadleaved Closed Forest	20095
Leaf Phenology	A3A10B2C1D1E2	Broadleaved Deciduous Forest	20097
2nd Layer: LF, C, H	A3A10B2C1D1E2F2F5F7G2	Multi-Layered Broadleaved Deciduous Forest	20628
3rd Layer: LF, C, H	A3A10B2C1D1E2F2F5F7G2 F2F5F10G2	Multi-Layered Broadleaved Deciduous Forest With Emergents	20630

The LCCS Translator Module provides the possibility to:

- translate existing legends into the LCCS language
- quantitatively assess the similarity of classes using LCCS as the reference base
- quantitatively compare classes at the level of the individual classifiers used

The Translator Module has been the focus of NMD NordLaM project work with the LCCS during 2002.

LCCS Translator Module work with Nordic class sets

Five Nordic landscape level mapping / monitoring legends were translated into LCCS terminology to examine the usefulness of LCCS as a bridging system and thematic harmonisation tool. A second objective was analysis of the types of problems encountered while translating classes used in different countries with similar landscapes but different approaches to land cover classification. This NordLaM project exercise is the first time that the LCCS Translator Module has been used in this way for a set of closely located countries.

The following five class sets were used; also shown below are the persons who made the translations of the classes in terms of the LCCS classifiers and attributes; in each case these were persons with significant knowledge and experience in the development and use of the specific class sets:

AIS-LCM, Denmark (Groom and Stjernholm 2001) - Geoff Groom, NERI (11 classes)

EELC, Estonia (Aaviksoo and Meiner, 2001) - Kiira Aaviksoo, EEIC (21 classes)

3Q, Norway (Fjellstad et al. 2001) - Wendy Fjellstad, NIJOS (57 classes)

DMK, Norway - Arnt-Kristian Gjertsen, NIJOS (8 classes)

SMD, Sweden (Ahlcrona et al. 2001) - Eva Ahlcrona, Metria Miljöanalys (53 classes)

Table 2 shows how three classes from these class sets were translated in LCCS.

Table 2. *LCCS translation of three of the 150 classes used in the NordLaM work with the LCCS Translator Module.*

Class from translated class set	LCCS Boolean formula
LCCS classifier or attribute : state used for this classifier or attribute	
1. Estonian land cover: "shingle and gravel coast"	B16-A5-A12-M214-M410
major category :	B16 (bare areas)
surface aspect :	A5 (bare soil a/o unconsolidated material)
qualifier :	A12 (stony) (5 - 40%)
environmental attribute :	M214 (lithology: gravel)
environmental attribute :	M410 (lithology : Quaternary: Holocene)
2. AIS LCM : "grass heath"	A12-A2-A11-B4-B13
major category :	A12 (semi-natural terrestrial vegetation)
life form of main strata:	A2 (herbaceous)
cover class of main strata :	A11 (open, between 25% - 65%)
height :	B4 (0.3 - 3.0 m)
qualifier :	B13 (0.03 - 0.3 m)
3. 3Q : "needle-leaved evergreen forest"	A12-A3-A10-B2-C1-D2-E1
major category :	A12 (semi-natural terrestrial vegetation)
life form of main strata :	A3 (trees)
cover class of main strata :	A10 (closed, > 75%)
height :	B2 (3.0 - >30.0 m)
spatial distribution :	C1 (continuous)
leaf type :	D2 (needleleaved)
leaf phenology :	E1 (evergreen)

Results of the NordLaM-LCCS Nordic class set exercise

Presented below is a summary of the main results from this exercise. A more detailed report and discussion from this exercise is presented in Jansen (in press).

a) Translation

i.e. how were the LCCS classifiers used to describe the classes from these five class sets?

Classes of the “Cultivated Area” (A12) major category:

The AIS and 3Q classes make use of a greater number of LCCS classifiers than the SMD and DMK classes, which mainly use only the <life form> classifier. (The EELC class set represented just semi-natural areas, being a sub-set of the full EEIC class set.) Certain LCCS classifiers are not used at all, eg. <field size>, <crop type>.

Classes of the “semi-natural vegetation” (A24, A24) major categories:

Use of the classifiers <life form> and <cover> is obligatory for describing a class in this category. For the classes of the five class sets, <height> is very widely used, <spatial distribution> is used infrequently and <leaf type> & <leaf phenology> are used occasionally. Interestingly the option of describing a class using the <stratification> classifier is not used.

Problems that were encountered in finding correspondence between original class criteria and LCCS classifiers are mainly related to:

- differences in the definition of threshold values
- the use of land-use terminology in legend classes
- original classes with values ranges that do not correspond to LCCS options ... resulting in (inappropriate) use of ‘mixed classes’ in LCCS

In addition, for these Nordic class sets the translation task was made difficult since the options in the current LCCS for description of lichen-dominated areas are very limited. For example, it is not possible to describe a class that is characterised by the presence of both lichen and other vegetation. Plans for revision of the LCCS (Antonio Di Gregorio, pers. comm. September 2002), include modifications of this part of the system.

As noted earlier, classification should relate to the current state of an object, rather than its past or future states. Several of the five class sets included classes for ‘burnt areas’ and ‘recently felled forest’, which presented particular translation problems since they imply aspects of either their past or future characteristics.

In general it was felt that translation would be easier and the result better if certain issues were discussed and agreed upon in advance. Also, guidance on use of LCCS classifiers, including good worked examples of class translation could benefit class set translation.

b) Similarity Assessment - legend intracomparison,

i.e. quantification of the similarity / distinctiveness of classes belonging to a single legend : AIS and EELC

Differences were noted between the AIS and the EELC legends in the distinctiveness of their classes belonging to single major categories. This could be associated with the greater frequency in the EELC class set of classes that are distinguished on the basis of a single classifier. Furthermore, these two class sets have different degrees of thematic detail, with just 11 classes in the AIS but 58 classes (21 used in this exercise) in the EELC.

It was apparent that intracomparison of a legend with the LCCS Translator Module can be a useful test for a pilot legend, to check for classes that may be difficult to separate in the mapping because they are indistinct from each other.

c) Similarity Assessment - legend intercomparison,

i.e. quantification of the similarity of classes in different legends : 3Q and SMD

As would be expected, the highest similarity scores are found between 3Q and SMD classes that both belong to the same one of the eight LCCS major categories. However, high similarity scores were also found between some 3Q classes and some SMD classes that belong to different major categories, in particular where these were the categories Cultivated Areas and (Semi-) Natural Vegetation. This cross-category similarity is associated with the occurrence of only a limited number of joint classifiers and the fact that these two main major categories are both classified in LCCS in terms of their plant life form and the vertical and horizontal arrangement of these plants.

It also has to be noted that the calculation of similarity scores in LCCS includes a bias in favour of comparisons in which the reference class (i.e. a class in one or other of the two class sets) is described in terms of only a few classifiers. Thus, comparison of classes against a reference class that uses just two LCCS classifiers can only result in similarity scores of either 0, 0.5 or 1.0 regardless of the number of classifiers used for the compared classes. This bias in the LCCS Translator Modules similarity scoring requires that it be resolved.

Conclusions

a) The NordLaM - LCCS exercise:

- Highly desirable as a good harmonisation of Nordic class sets related to the mapping and monitoring of natural areas would be, it was not the objective, or the outcome of this exercise to achieve this.
- However, it provided important experience in use of a systematic standardised classification system and also insight into the Nordic class sets used.
- Several areas for improvement and development in LCCS were noted:
 - A class set that has been translated and input to LCCs as an LCCS legend currently requires to be re-entered class-by-class into the Translator Module. This entry of legends to the Translator Module should be made more automated, and the interface for this task made simpler.
 - The need for further consideration of the LCCS TM class similarity score algorithm
 - The need for further development of the LCCS concept for lichen dominated areas. Given the major knowledge and experience of these areas, on account of their extents in Nordic countries, the Nordic community should consider collaboration with the LCCS on this topic.

b) Class harmonisation / correspondence in general:

- A systematic classification system approach, such as LCCS is important, even obligatory, for high quality legends
- An approach to classification that only creates a legend, however sophisticated and thought-through, is inadequate.
- The LCCS represents a significant step towards more robust classification for natural areas mapping and monitoring, and represents an important bridging tool for making correspondence between land related class sets.
- Further consideration is required on the role of a land cover orientated classification for broader natural areas mapping and monitoring work.

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Using the preceeding interglacials to provide a dynamic base-line for European Nature

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Motivation: the need for a base-line for nature in northwestern Europe

The succession threat

Among the most important problems facing nature conservation in Europe as well as world-wide is habitat loss and the concomitant threat of species extinction. In north-western Europe several factors are presently causing habitat loss, notably land-use changes and eutrophication (e.g., Stoltze & Pihl 1998). One of the predominant processes resulting from the change from the 'traditional' extensive agriculture typical of the pre-industrial landscape to the modern landscape is secondary succession towards scrub and forest in areas of marginal interest to modern agriculture and in protected natural areas. This secondary succession causes the loss of open- and semi-open habitats and is among the major negative factors impacting on the redlisted species in e.g. Denmark (Stoltze & Pihl 1998). Consequently, secondary succession and the loss of open habitats is generally seen as a biodiversity threat (e.g., discussion in Vera (2000)). However, this interpretation depends on our base-line for nature.

Why not the pre-industrial cultural landscape as a base-line?

The dominant approach in north-western European nature management has been to explicitly or implicitly use the conditions of the 'traditional' cultural landscape (TCL) as the base-line for conservation status. Often it has taken the form of simply trying to maintain the conditions at the onset of monitoring or establishment of a given nature reserve. The advantage of this approach is that we often have direct, good-quality information about the base-line and the processes that were involved in maintaining the base-line conditions. The TCL approach however has some serious shortcomings:

- The TCL base-line is an arbitrary and unique point in the postglacial development of a human-dominated landscape. There is no compelling biological reason why this particular point in time has been chosen, rather than any other such as the late Neolithic or the Iron Age.
- The TCL base-line is meaningless when viewed from the temporal perspective over which present-day European biodiversity has developed (e.g., Svenning 1999). The far majority of European species most likely have arisen and had most of their evolution during the Pleistocene or even earlier (e.g., Nilsson 1983; Lang 1994; Hewitt 2000). If the nemoral open- and semi-open habitats in northwestern Europe have no history beyond the, from an evolutionary point of view very brief, period of agriculture, secondary succession can be interpreted as a process favouring native woodland species at the expense of 'exotic' species, which only invaded following the anthropogenic destruction of the 'wildwood'. In this case, secondary succession could hardly be argued to be a threat to global biodiversity.
- Finally, the TCL base-line is a static one and consequently will be increasingly impossible to maintain as time passes, in particular due to the ever-changing, naturally and now probably strongly enhanced by humans, climate (cf. e.g., Sykes et al. 1996).

Hence, it seems clear that we need to find a replacement for the TCL base-line for European nature.

The preceding interglacials can provide a more relevant and dynamic base-line

One logical reference point for European nature is its present-natural state (sensu Peterken 1996), i.e., the state it would have been in today had humans never been present. However, given that no larger lowland area in Europe remains in a truly virgin state (e.g., Peterken 1996; Vera 2000), how can this concept be operationalized? In Svenning (2002) I have advocated one approach, which will be briefly summarized here. During the last 900,000 years the climate of north-western Europe has oscillated in a complex manner between arctic and temperate conditions. The present interglacial, the Holocene, is characterized by an oceanic, warm climate. Given the comparable climate and minimal human impact (modern humans absent), the vegetation of the temperate phase of the preceding oceanic Middle and Late Pleistocene interglacials (OPI) can be used as analogues for the present-natural vegetation (the OPI base-line). It needs to be emphasized that the point of the OPI base-line is not to replicate exact conditions of a particular past period, but to use knowledge of the past to get the best possible estimate of the present-natural conditions. The main advantages of the OPI base-line are the following:

- Refers to a time-scale much more relevant to the evolution of the present-day European biodiversity.
- Provides a true base-line for the natural conditions (no or at least minimal human impact), rather than an arbitrary level of human impact in ever-evolving human-nature interaction. Although some human impact on fauna and vegetation during the Pleistocene interglacials cannot be excluded, the effects have surely been small compared to the Holocene. Notably, in Europe the end-Pleistocene megafauna extinction event took place after the Eemian interglacial (Stuart 1991).
- Provides a dynamic base-line because it should be interpreted in a process-oriented fashion to derive the present-natural conditions, i.e. taking natural or unavoidable climatic and edaphic changes into account.

The main shortcoming of using the OPI base-line is the uncertainties involved in establishing ecological conditions and processes so far back in time. However, the science of palaeoecology becomes ever more apt at providing such information and already today provides information of sufficient quality for a practical development of the OPI base-line (Svenning 2002).

Methods: Review of palaeoecological data

Recently, I attempted to provide an OPI base-line for Northwestern Europe (north of the Alps to southern Sweden, west to the British Isles and east to eastern Germany) by examining the palaeoecological literature regarding the vegetation during the preceding oceanic interglacials, supplemented with some studies from the pre-agricultural part of the Holocene (Svenning 2002).

Results: The OPI base-line for the northwestern European nature

Below, I briefly summarise the findings of Svenning (2002).

Habitat types and species diversity

The palaeoecological evidence from past interglacials and pre-agricultural Holocene allowed me to make the following conclusions regarding the types of terrestrial habitats likely to be present in north-western Europe today in the absence of human influence (Svenning 2002):

- The vegetation in uplands with more or less fertile soils would predominantly be closed forest, but with localised longer-lasting openings, e.g. around ponds and probably also grassy glades. Despite general shade tree dominance, oak, hazel, and often also Scots pine would be able to maintain themselves, albeit at variable densities, in the forest vegetation

throughout north-western Europe, contrary to the present situation reported from many nature reserves (Vera 2000).

- Open vegetation would mainly be present on floodplains and on highly calcareous or oligotrophic soils (as well as in the most continental parts of north-western Europe). The dominant vegetation on floodplains would frequently be a mixture of open marshes and meadows, disturbed ground, dry grasslands, scrub, and some woodland or forest. Calcareous grassland would also have at least some presence in chalkland uplands, and the vegetation on infertile soils would generally consist of fairly open woodland as well as patches of heath and grassland.
- Open habitats would be sufficiently represented to allow for the existence of many species of open woodland, scrub, woodland glades, dry acidic and calcareous grassland, heaths, meadows, open marshes, and bare, disturbed ground. Given the palaeoecological evidence it is probable that a major proportion of the species found in the traditional cultural landscape would also occur in north-western Europe under present-natural conditions.

Diversity-maintaining processes

The palaeoecological evidence on the processes involved in maintaining the documented habitat and species diversity is less clear. However, at least two key processes are suggested (Svenning 2002):

- The interglacial large herbivore fauna of nemoral Europe was highly diverse, including not only the present-day types of wild herbivores such as wild boar, beaver, various deer, and bison, but also aurochs, wild horse, and even elephants and rhinos. Particularly the abundant and diverse dung beetle fauna often found in floodplain deposits suggests that these large herbivores were the main agents maintaining open and semi-open habitats on floodplains (and by inference probably also of some importance in other landscape types). Other authors have also argued for key role of large herbivores in the maintenance of habitat and species diversity in boreonemoral and nemoral Europe under natural circumstances (e.g. Andersson & Appelquist 1990; Vera 2000).
- Records of charcoal and charred wood suggests that fires may, at least sometimes, have been a key factor in maintaining the oaks, hazel, and pine in upland forests (although other factors such as variable hydrological conditions, wind-throws, and perhaps localized herbivore activities were probably also involved) and in maintaining open and semi-open vegetation on poor soils.
- Overall, a picture is emerging where climatic and edaphic-topographic factors would interact with herbivores, fires, and probably also other disturbance factors to produce a diversity of forested and open habitats under present-natural conditions.

A changing world

Not specifically discussed in Svenning (2002), another feature of the palaeoecological records of profound importance for the OPI base-line is the near ubiquity of climatically and edaphically driven temporal changes in the habitat and species composition (e.g., Nilsson 1983; Iversen 1958). While the climatic vagaries of the Pleistocene have caused numerous losses from the European flora (Svenning 2003), the present-day flora and fauna has survived and evolved through this period (e.g., Nilsson 1983; Lang 1994).

Discussion: Management implications

Reinstall diversity maintaining processes

Apart from the fact that closed forest would indeed predominate in much of the landscape under present-natural conditions, the present problem with secondary succession also reflects that key

natural processes in the maintenance of habitat diversity are severely constrained in the modern landscape of north-western Europe. As indicated by the palaeoecological record these include the grazing, browsing, and trampling of large herbivores and the occurrence of natural fires (Svenning 2002). Another factor of likely importance is a variable hydrological regime (Buchwald 2001), again strongly constrained due to widespread draining. Free-ranging large herbivores would also be important for the maintenance of species diversity by providing long-distance dispersal for many herbaceous plants. The present limited occurrence of free-ranging wild or domestic herbivores probably has as consequence a dramatically decreased chance of dispersal for most plant species in the modern landscape as compared to both the traditional cultural landscape and the OPI base-line (Bruun & Fritzboøger 2002; Poschlod & Bonn 1998). The likely consequence is decreasing species richness in terrestrial plant communities such as dry grasslands, meadows, and heathlands (Bruun & Fritzboøger 2002). Consequently, it must be a conservation priority to re-establish natural diversity-maintaining processes wherever possible

Do not forget "missing" habitats

While Svenning (2002) supports the significant representation of open habitats under present-natural conditions, it is important to note that the most widespread natural vegetation type would be closed old-growth forest and that scattered old trees and dead wood probably also would be present in the more open habitats. Many organisms dependent on old-growth forest, old trees, or dead wood have not survived well in the cultural landscape (see references in Svenning 2002). Thus, ancient forests should be protected and the presence of old trees and dead wood promoted in most habitats. Further, as discussed by Svenning (2002) large herbivores and fires would provide the special microhabitats needed by a range of dung- and fire-dependent species, many of which have become rare or locally extinct in north-western Europe (also cf. Andersson & Appelquist 1990). I.e., a further argument for a renewed role of herbivores and fire in the European landscape.

The necessity of a dynamic, large-scale approach

At local or national scales an approach focused on maintaining status quo in natural areas will be doomed to failure in the face of natural and more urgently human-caused climatic change. Consequently, there is a need for the development of a continental-scale, dynamic approach, something the OPI base-line concept is well suited to be part of (cf. Soulé & Terborgh (1999) for a remarkable contribution to such an approach for North America). Apart from providing a base-line that can be used even in the face of climatic change, the OPI approach may also help identify factors important in mitigating the negative impacts of climate change. Notably, by providing efficient long-distance dispersal free-ranging large herbivores may well be crucial for the ability of many plants to track climatic changes.

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Favourable conservation status in forests and its indicators

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Introduction

The indicator paradox

The use of indicators in nature management is an attempt to reduce complexity in nature, in order to gain relevant and operational monitoring tools. This reduction will invariably conflict with the parallel wish to get a representative picture of the complex reality. Good indicators bridge this gap in an optical way, securing a balance between reduction and representativity. In addition the good indicator needs to be easy to monitor and should be highly reproducible.

Defining the favourable

Before optimal indicators can be identified it is necessary to define what kind of process or state they should indicate, and in which context they are to be used. In this mini-essay, I will focus on potential indicators of habitat quality at stand level in nemoral, deciduous forests. I define a favourable conservation status as one securing the maintenance of vital populations of the majority of organisms regarded as native to, but threatened within a given area, with the highest responsibility for endemic species or species with their main distribution within the area. To focus on very simple biodiversity measures, e.g. species richness, seems to me a too simple approach – for two reasons. Firstly, because species richness is highly variable among habitat types and, secondly, since a high species richness may be maintained locally by the replacement of native species with a restricted distribution area by introduced species with a wide distribution area, leading to a decrease in regional or global bio-diversity.

In this context, I also need to add that I find it essential to focus on all taxa, including fungi, insects and microorganisms, rather than just the cute and well known; plants, mammals and birds. The UN Convention on Biological Diversity of 5 June 1992 gives a such ethical responsibility, but even by considering homocentric perspectives such as usability and ecosystem functioning this seems wise, given the enormous diversity in ecological roles, metabolite production etc. found among microorganisms (e.g. Spear 2000). An obvious problem in this context is that we know little on microbial diversity in Nordic forest ecosystems, the factors influencing it and how it is related to macrobiotic diversity. Studies in this field are badly needed.

With a focus on threatened organisms, well-documented red-data books provide essential management guidelines. In this context, red-data books covering habitat types would be a valuable development. Guidelines given in Natura 2000 can be seen as a step in this direction in the European perspective, but they seem in many cases to be too imprecise and general to catch up with essential qualities at the regional scale.

Habitats, structures, continuity

The problem of co-variation

Forests are complex, habitat- and process rich ecosystems, differentiated both on the vertical and horizontal scale. Typically, threatened organisms depend on certain habitat qualities, of which some appear to be especially attractive for a large number of species. Thus, many lichen

species prefer a humid forest climate, whereas many wood insects prefer warm, sun exposed snags and logs. Though both qualities may be combined in a single site, often they are not. Even in relatively simple ecosystems, like grasslands, studies have shown that different taxonomical groups often are attracted by different habitat qualities. Accordingly, species richness between groups may be completely uncorrelated (e.g. Vessby et al. 2002). The few studies conducted in boreal forests have given similar results (Ohlson et al. 1997, Jonsson & Jonsell 1999) and there is no doubt, and some evidence (Nilsson et al. 1995), that the same is true in the nemoral forest zone. In management of forest biodiversity it is essential to respect this complexity. Preferably, by attaining a multi habitat approach in management and monitoring plans, or less ambitiously, by clearly stating, which aspects of bio-diversity that are considered.

Habitats

Forest habitats or elements, which I identify as especially important in nature management in the nemoral zone are:

- Dead wood
- Veteran trees
- Certain soil and humus types
- Humid forest climate
- Water (forest swamps, streams)

“Certain soil or humus types”, denote special soil qualities related to low levels of disturbance and/or the presence of certain bedrock types. It is well known that calcareous soils often support unique plant and fungal communities, but also other aspects of soil and litter quality have importance. For instance, certain communities of soil fungi, mites and nematodes seem to be restricted to old growth forest stands, characterised by high continuity in soil- and litter conditions (Heilmann-Clausen et al. 1992, Møller 1997, Bjørnlund et al. 2002).

The importance for bio-diversity and/or parallel decrease of the other listed habitat elements have been reported elsewhere (e.g. Söchting 1992, Berg et al. 1994, Rune 1997, Fridman & Walheim 2000, Møller 2000, Siitonen 2001, Nilsson et al. 2002), for which reason I will not go in further detail with each element here. Instead I will try to put each element into a bigger context.

In figure 1, a larger number of typical forest habitat elements are ordered in a two dimensional space defined by habitat persistence in time on one axis, and habitat predictability in space on the other, according to a presumed present natural situation. My ordering of habitat types is a generalization, and is mainly meant to give an impression on how ecological forces are likely to influence dispersal strategies among organisms associated with various habitat types. For instance a “moist microclimate” takes up a rather extreme position, being characterized by high persistence and predictability, since I imply that a moist microclimate will be prevailing in natural nemoral forests. On the other hand, I regard burnt wood and humus as a highly unpredictable and transitory element since forest fires must be regarded a rare and local phenomenon under present natural conditions in the nemoral zone (Niklasson & Nilsson 2002).

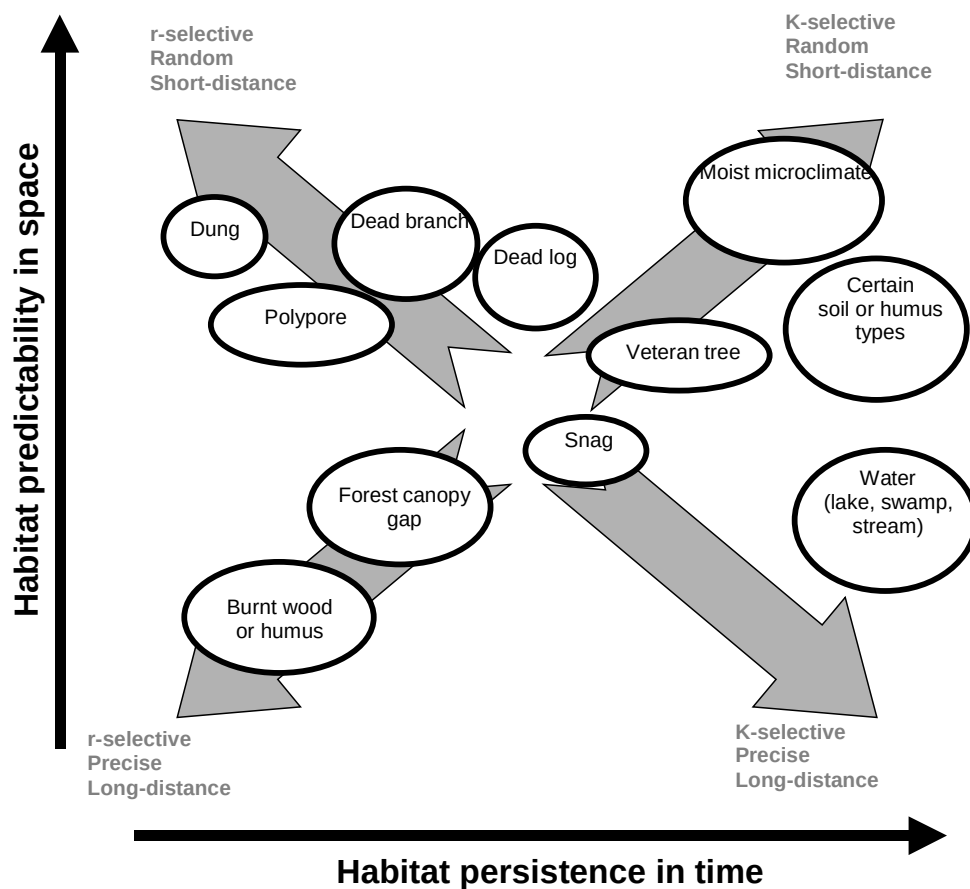


Figure 1. Persistence in time and predictability in space of various habitat elements in a presumed natural, nemoral deciduous forest. The arrows indicate presumed optimal reproduction and dispersal strategies. The placement of habitats in the diagram is a generalisation.

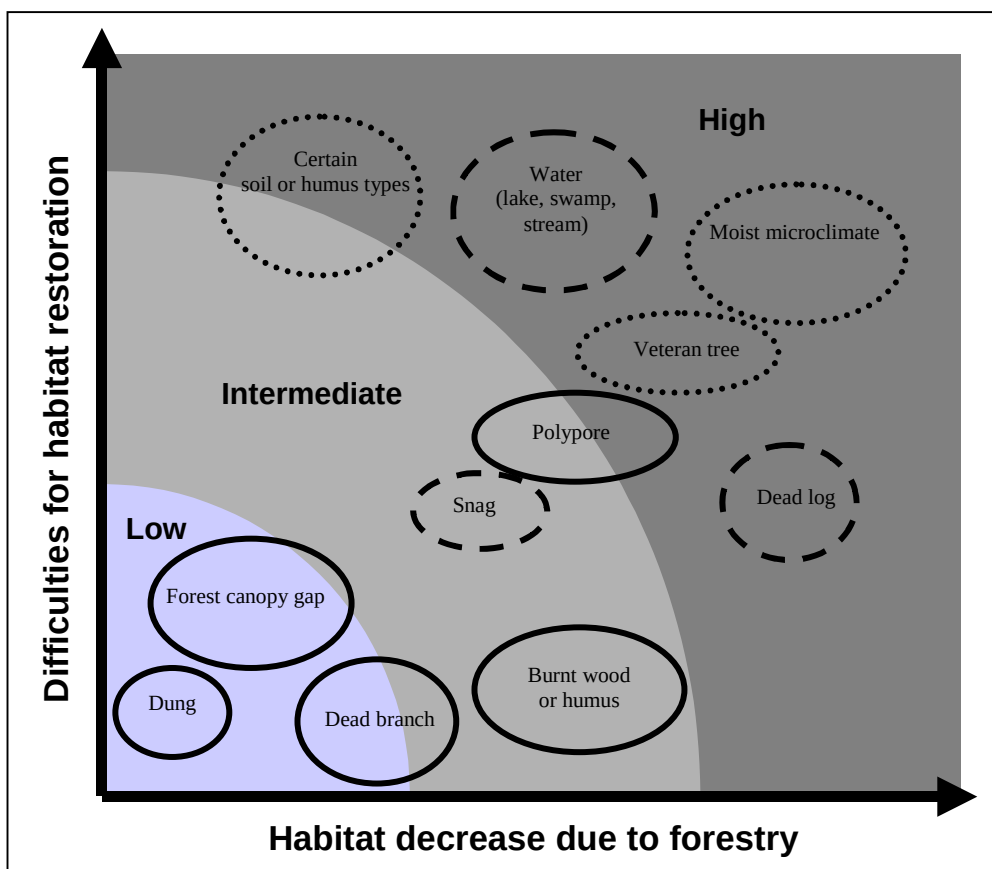


Figure 2. The presumed effects of forestry on the habitat elements in figure 1 with an indication of difficulties for habitat restoration. The grey zones indicate theoretical levels of vulnerability based on the combination of axes 1 and 2. Finely punctuated ellipses indicate habitats sensitive to forest fragmentation, due to selectivity for low reproduction rates and random, short-distance dispersal; coarsely stippled ellipses indicate less sensible habitats, both based on Figure 1. The placement of habitats in the diagram is a generalisation.

In figure 2, the same habitat types are considered in a management context, ordered in a two-dimensional space defined by habitat sensibility to traditional forestry practices on one axis, and habitat restorability on the other. Again, it is important to note that management practices and their impact on various habitat types differ widely in the nemoral forest zone. My approach is a generalization, referring to the general Danish situation.

The inclusion of theoretical dispersal strategies from figure 1, shows that several of the most management sensitive habitat types are, at the same time, selective for random, short-distance dispersal, thereby adding to the vulnerability of the associated organisms in a fragmented landscape.

Structures

In general, managed nemoral forests are very differently structured compared to natural forests. Most management practices are based on rather large (one-several ha) even-aged, monoculture units, which form a coarse grained spatial mosaic. Natural stands are, on the contrary, typically composed of numerous, small (100-1500 m²) comparably heterogeneous units, which form part of a shifting mosaic (Christensen & Emborg 1996, Emborg et al. 2000). The precise impact of this aspect of forestry on bio-diversity is unknown, but it seems likely that species depending on short distances between different forest development phases or live in the borders between these are negatively affected (Christensen & Emborg 1996). This may be the case e.g. for birds, insects and small mammals, which during one day - or the year - are dependent on several habitat types for survival, e.g. hollow trees for nesting and forest canopy gaps for hunting insects or eating fruit. Also at the vertical scale, structural diversity tends to be lower in managed forests. The effect on bio-diversity is probably limited, but multilayered natural stands may have a more humid microclimate, compared to managed stands, which may benefit lichens, bryophytes and other organisms sensitive to desiccation.

At a larger scale, landscape connectivity is very important for animals with large territories, e.g. larger carnivores as well as for lichens and fungi, which depend on random, long distance spore-dispersal. Also the mingling of forest with non-forest habitats, such as lakes, swamps and grasslands is an important aspect. In general, drainage and intensified forest management had led to a decrease, both in the number and size of such habitats in nemoral forests (e.g. Rune 1997). Again, research is scanty, but it seems likely that the decline in structural diversity has led to a substantial decrease in bio-diversity in most managed forest landscapes. Summing up, an increased focus on improving structural diversity at landscape and forest level seems to be important, if guided by relevant research initiatives.

Continuity

The representation of habitats and structures in a landscape or at a single site may be stable in time, or highly fluctuating due to natural disturbance regimes or human influence. Large past fluctuations may have led to the extinction or exclusion of specialised organisms depending on continuity, i.e. a highly stable representation of specific habitat qualities. On the other hand, future imbalances may lead to failure in the attempt to preserve such specialised organisms. In Denmark a number of forest reserves have been designated in the 1990'ies, in many cases to protect valuable, isolated, oldgrowth stands (Skov- og Naturstyrelsen 1997). Unfortunately, many of these reserves are fairly even aged, making it easy to predict that they, ultimately, will collapse and go through a regrowth phase unsuitable for most organisms associated with oldgrowth forest. If such areas, and this often the case, are not enlarged by including younger, at present less ecologically valuable stands, the conservation strategy must be denoted as non optimal and unfavourable.

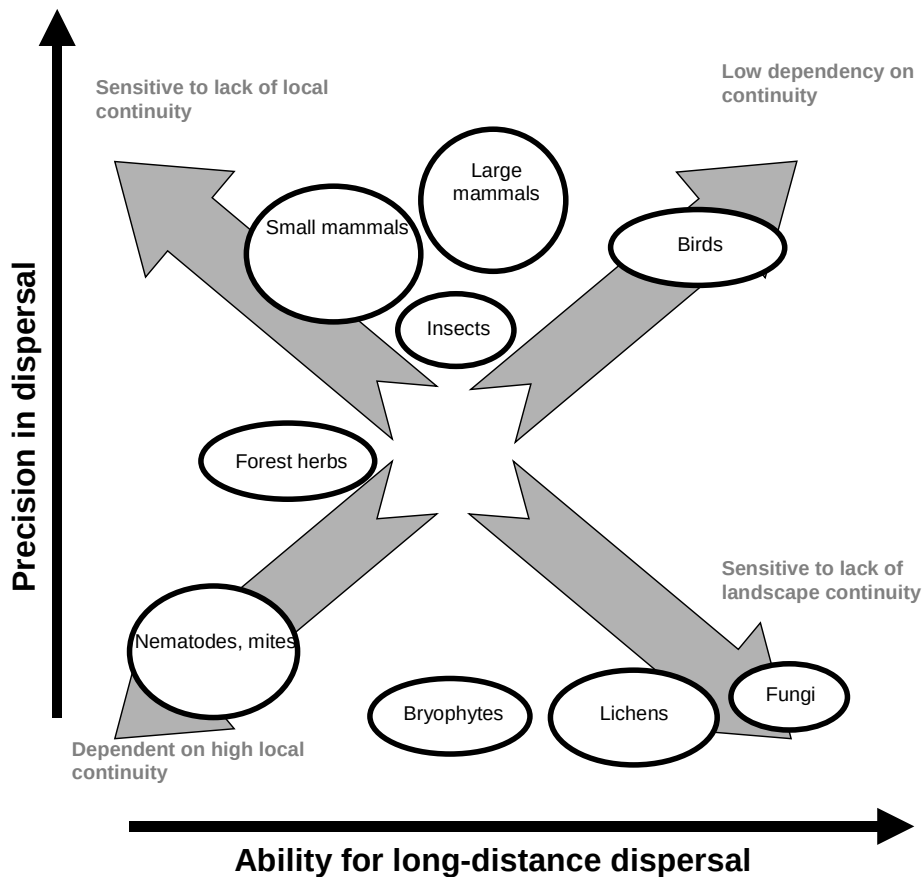


Figure 3. Various organism groups and their dispersal abilities in the nemoral forest zone. Arrows indicate theoretical dependency of habitat continuity on local or landscape scale, based on combining axes 1 and 2. The placement of each species group in the diagram is a generalisation, covering large differences within several groups; most notably insects.

It is thus important to consider both the aspect of past continuity and that of future dynamic potential, when selecting and managing forests for bio-diversity, and both aspects need to be encountered at the appropriate scale and habitat level. For organisms, e.g. forest herbs, soil mites and beetles depending on hollow trees, relying on short distance dispersal and the presence of predictable habitats, local continuity is paramount (cf. Nilsson & Baranowski 1997, Ohlson et al. 1997). On the other hand, organisms with good or intermediate dispersal ability and association with temporary habitats, e.g. dead wood, are likely to be indifferent to local continuity, but highly dependent on continuity at landscape scale (Figure 3). Similarly, at the habitat type level, plants and epiphytic lichens are independent on continuity in dead wood, whereas wood-inhabiting fungi and insects appear to be fairly independent on the continuity in forest microclimate and soil conditions.

Structural indicators versus indicator species

A wide range of indicators has been suggested and to some degree implemented for monitoring forest habitat quality at stand level (e.g. Haugset et al. 1996, Kotiranta & Niemelä 1996, Møller 1997, Ferris & Humphrey 1999, Nitare 2000, Larsson 2001, Newton & Kapos 2002). The suggested indicators can be separated in structural indicators, directed towards registration of key habitat parameters and indicator species, directed towards the approach of monitoring species assumed to be representative for whole assemblages of species or species diversity in general.

Structural indicators

Structural indicators have several advantages for inclusion in monitoring programmes. They are often easy to understand for foresters or other technical staff, which traditionally perform forest

monitoring, and are thus easy to include in existing programmes. In addition, structural indicators are very suitable for modelling and can thus be used to make predictions on future development in bio-diversity under various management scenarios.

On the other hand, structural indicators represent an entirely indirect approach: We are only rarely interested in conserving habitats *per se*, rather in conserving the associated species. This means that the links between chosen structural indicators and conservation goals (e.g. preserving of threatened species) need to be very well supported. In the boreal part of Fennoscandia substantial work has been performed to examine the strength of such links (e.g. Ohlsson et al. 1997, Esseen et al. 1999, Gustafsson et al. 1999, Jonsson & Kruys 2001). In nemoral Europe such work is still scanty (but see Aude & Poulsen 2000, Ranius & Jansson 2000, Berg et al. 2002, Christensen & Heilmann-Clausen 2002, Heilmann-Clausen & Christensen 2003) and is badly needed before the relevance of structural indicators can be justified.

Another drawback of structural indicators is that some key features are very difficult to account for. This is e.g. the case for forest climate, which is very laborious to monitor. The same is the case for stand continuity, which is the subject of debate these years (Ohlsson et al. 1997, Nordén & Appelquist 2000, Groven et al. 2002, Rolstad et al. 2002). Though I agree with these authors that the role of stand continuity has been overemphasized, not least in the boreal zone, I think forest continuity, in various habitats, and at various scales (not least the landscape scale), is very important in the nemoral zone, characterized by an earlier and much more thorough fragmentation of the forest landscape, compared to most of the boreal parts of Scandinavia and Russia.

Indicator species

Several indicator species concepts exist and a range of more or less confusing terms, (e.g. umbrella species, focal species, flagship species) has been suggested by various authors often leading to confusion rather than enlightenment, though the original concepts are well defined (cf. Caro 2000). Here, I will not go in debt with existing concepts (but see Larsson 2001) but refer directly to how the concept of indicator species have been used in connection to habitat quality in Nordic forests, without paying too much attention to the range of available terms. Indicator species are appealing for the interested amateur and have been used rather extensively in Norway and Sweden as a measure to identify so-called key-habitats, valuable for the conservation of threatened organisms (Hansson 2001). In regular forest inventories indicator species have yet to gain popularity and some serious struggles need to be taken, before this situation will eventually change. Indicator species do typically require some level of specific knowledge, beyond the educational background of most forest practitioners involved with forest inventories. In addition, indicator species may be more time demanding to monitor in the field compared to simple habitat parameters. Another serious problem is that in most cases indicator species have been suggested and promoted based on qualified guesses by experienced field biologist (my self including!), without any testing and with a rather loose concept. In other words the indicator species do indicate "valuable habitats or conditions", without a precise definition of what is valuable, and the attempt to justify the quality of each indicator species is lacking.

However, some attempts have been made to test the value of indicator species, following two main approaches. Jonsson & Jonsell (1999) and Sjøgren-Gulve (1999) promoted the "nested species subset hypothesis" in the Nordic context. The hypothesis implies that a set of species form nested subset patterns if species that occur on sites with n species also tend to occur on sites with $n+1$ species. In the presence of nested subset patterns species with intermediate or rather low numbers of occurrences are accordingly suitable as indicator of species of species rich sites, where rare species are most likely to occur.

An alternative approach has been to test the ability of actually suggested or indicator species to actually indicate something, typically the presence of threatened species, within a functional or taxonomical group of organisms (e.g. Nilsson et al. 1995, Kuusinen 1996, Ranius 2002). To me, the nested subset hypothesis is unsatisfactory and highly reductionistic. Firstly, it builds on the assumption that species richness within a given species group varies uniformly, in response to a single habitat quality (or size) gradient. Deviation from nested subset patterns is interpreted as randomness in the distribution of species across sites, in which case the use of indicator species is regarded as problematic (Jonsson & Jonsell 1999). The hypothesis does thus not account for situations, where species richness depend on two or more non-correlated gradients, which I think is often the case, even for species groups with presumably rather uniform habitat demands, e.g. epiphytic lichens. Secondly, but related to the first aspect, the approach focuses entirely on species richness and does totally neglect the fact that rare or threatened species may be rare or threatened because they depend on special habitat qualities to which more common species are insensitive. Accordingly, species richness and the presence of threatened organisms are not necessarily correlated. Finally, if nested subset patterns are eventually found, the objectively found indicators are quite likely to be unsuitable as indicators, due to inconspicuity, taxonomical problems etc. hindering a successful implementation in practice.

The alternative, less rigoristic approach, is thus preferable. It combines, in a quite optimal way, qualified guesses and intuition of experienced field biologist with objective testing procedures and can be focussed on species with obvious indicator potential. However, the nested subset approach may be applied in parallel for well-defined, isolated situations, e.g. to find optimal indicators among a larger subset of suggested indicator species and/or with focus shifted from species richness to richness of threatened species.

Structural indicators, indicator species - or both?

The question is now whether the focus should be on structural indicators or indicator species when monitoring forest bio-diversity? My answer is both – at least at present! Structural indicators have an obvious potential in relation to rather simple habitat elements such as dead wood, water in the forest etc., but seem inappropriate with respect to more complex or less understood features such as microclimate, old growth soil qualities and landscape continuity. In both cases it is essential with more development and research if monitoring programs are to be successful and cost-effective. An understanding of nemoral forests as very complex habitats and the identification of all habitat elements, structures and processes necessary for an efficient management of biodiversity, is essential in this process. Conservation of forests as complex ecosystems is haphazard if a too simplified indicator system is used. I will recommend the following procedure:

First of all, the definition of what is considered a “favourable conservation status” needs to be absolutely clear, and this is essentially a political responsibility, though scientist may guide the process. Local, regional, national and over-national authorities shall give clear and concise guidelines at each level. In the hand of conservation biologists the question typically becomes muddled due to expert discourse on which taxa, habitats, structures etc. that are most important, often rooted in deviating personal and/or financial interests.

When guidelines are clear, the next step is to identify habitat elements, structures and processes that are essential in order to realize the conservation goal given by the guidelines and the economical resources available. Subsequently the task is to identify optimal indicators for each essential habitat parameter. At first, by testing already suggested and well-defined, highly reproductive indicators - species or structures - according to their ability to indicate identified

habitat qualities, and if this fails, by screening relevant new indicators selected on the basis of common, biological sense.

Finally, the selected indicators need to be combined in to a practical and operational system. Ferris & Humphrey (1999) and Nilsson & Niklasson (2002) gives an impression on how this can be done, but the issue can no doubt be further developed. For the other steps in the process, it is my hope that the present contribution can be of some help.

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Numerical methods for evaluation of biological condition in open land habitats

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Abstract

This paper motivates the need for methods to obtain data-based evaluations of conservation status and trend of terrestrial habitats. The paper describes a method for statistical evaluation of conservation status of terrestrial habitats from inventories of vegetation. The method is based on classification modelling (pattern recognition) of a priori classes selected within reference data sets that reflect ecological gradients of importance to conservation status. A Habitat Quality Model is described that discriminates between culturally improved grasslands and meadows and natural and semi-natural grasslands, heaths, dunes, fens, mires and meadows. Validation and evaluation of the model revealed that it effectively identified habitats of high conservation value. Unfortunately, early successional stages of infertile habitats (e.g. abandoned fields) were also placed in the high value category by the model.

This motivated the development of a Succession Model for discrimination between old semi-natural grassland and heathland and abandoned fields. When the two models are combined, we get a strong indication of the conservation value of grassland and heathland vegetation.

Statistical and ecological assumptions and prerequisites for successful model construction are discussed and perspectives for future applications are outlined. Because such quality assessment models are based on species composition, we recommend that they be combined with evaluations based on population trends of rare or vulnerable species as well as measurements of physical and chemical conditions of the habitats.

Introduction

The demands of the EC-Habitats Directive (Anon. 1992) are a challenge for numerical ecology. Favourable conservation status stands as the central concept. Favourable conservation status is defined as the fulfilment of three major criteria. According to article 1 (e) of the directive, the conservation status of a natural habitat will be taken as 'favourable' when:

- its natural range and areas it covers within that range are stable or increasing, and
- the specific structure and functions which are necessary for its long-term maintenance exist and are likely to continue to exist for the foreseeable future, and
- the conservation status of its typical species is favourable as defined in (i);

And according to article 1 (i) the conservation status of a species is taken as 'favourable' when:

- population dynamics data on the species concerned indicate that it is maintaining itself on a long-term basis as a viable component of its natural habitats, and
- the natural range of the species is neither being reduced nor is likely to be reduced for the foreseeable future, and
- there is, and will probably continue to be, a sufficiently large habitat to maintain its populations on a long-term basis.

The Habitats Directive call for status reports every 6 years includes a type-specific information on trends in favourable conservation status. A certain level of type-specific mapping and monitoring inevitable. Furthermore, the concept of favourable conservation status needs to be

developed in order to be operational: What set of biotic and environmental parameters are appropriate for assessing conservation status and the development in conservation status over time? A prerequisite for successful monitoring is further that the collection of parameters and the subsequent evaluation of data can be carried out in a standardised and reproducible way.

The objective of the applications presented in this paper was to develop methods for statistical discrimination between open land habitats of high conservation value (with special emphasis on grasslands) and cultural habitats of low conservation value.

The evaluation concept

Reference data

The approach of the following applications is based on establishment of a reference data set that reflects the species composition along relevant environmental gradients. Relevant environmental gradients are defined here as gradients that reflect identified threats to the habitat types in question, e.g. drainage or eutrophication. The reference data set should be sufficiently large to mirror the natural variation in species composition within the geographic area in question along the environmental gradients. Depending on the number of relevant environmental gradients, it may be necessary to establish more than one reference data set.

Species composition

Species composition is, unlike species richness, not a single continuous variable. It is therefore difficult to use raw species composition data in statistical models. Ordination, i.e. the extraction of floristical gradients – so-called coenoclines, is one way to circumvent this difficulty (see details below). Ordination reduces the dimensionality of species data to a low-dimensional space reflecting a major part of the deterministic variation (Ejrnæs, 2000). It is important to notice that gradients extracted by ordination do not reflect species richness, but species composition. The coenoclines extracted through ordination come in the form of continuous ordination scores, i.e. sample and/or species co-ordinates in an n-dimensional space. These scores are used in statistical modelling. When a reference ordination has been established, the co-ordinates of new samples may be calculated by passive ordination.

Training data

Within each reference data set, a training data set needs to be extracted. The training data set consists of samples that, additional to the species information, have predefined conservation status. The information on conservation status may take the form of two or more a priori classes, e.g. “favourable” and “unfavourable” conservation status or it may take the form of a continuous index value reflecting conservation value.

The classifier

The classifier represents a statistical model that predicts the conservation status from the sample co-ordinates in ordination space. A classifier may be constructed by different statistical methods, and in the case of a training data set with classes, these may be e.g. logistic regression (in cases with only two a priori classes), multinomial regression, discriminant analysis, neural networks or classification trees. The classifier is constructed on the training data set, and may be used to predict the conservation status on other data.

A recurrent challenge in statistical modelling is to avoid the over-interpretation of data – not least when flexible non-linear or “black box” methods are used. Parsimonious models may be achieved with the use of validation techniques – e.g. cross-validation (see below). It is also wise to test the properties of the model on different types of test data, chosen specifically to answer questions such as:

Is the model capable of identifying all kinds of habitats with favourable conservation status (false negatives)?

Is the model correctly identifying all kinds of habitats with unfavourable conservation status (false positives)?

How robust is the model to sampling method (e.g. plot size)?

Applications

We will present two applications of such classification models with relevance to Danish vegetation. The first model, the classification model, aims at a discrimination between the low habitat quality of drained and improved grasslands, meadows and pastures on one side, and natural and semi-natural dunes, heaths, grasslands, fens, mires and meadows on the other side (Ejrnæs et al. 2002). The second, the succession model, aims at a discrimination between abandoned fields on one side, and semi-natural grasslands and heaths on the other.

The habitat quality model

For the establishment of a reference data set, data representing the present state of the Danish countryside was combined with data from reference-areas representing a minimum of agricultural pressures on conservation value. A training data set was classified into two classes corresponding to low and high influence of agriculture.

Data for the study derive from a 1998-inventory and from DANVEG, a database of published and unpublished accounts of Danish – mainly natural, and semi-natural – terrestrial vegetation. These data derive from a large number of separate studies. While all studies imply samples from homogeneous vegetation, sample plot sizes vary considerably – from 1 m² up to approximately 100 m² (Nygaard et al. 1999).

The 1998-inventory was carried out in 10 transects of 1 x 5 km, distributed along a regional stratification from naturally fertile eastern Denmark to naturally infertile western Denmark. Stratified random sampling was used with a plot size of 49 m² (consult Ejrnæs et al. 2002 for more details).

Two subsets of the DANVEG database were selected. One subset was selected to represent the variation in semi-natural and natural, open terrestrial plant communities of Denmark (subset = semi-natural). This subset was obtained by random drawing of 10 vegetation samples from all plant communities belonging to dunes, fens, mires, bogs, heathlands and dry grasslands. Samples without cryptogams were omitted leaving us with 333 samples.

A second subset was defined as samples containing plant species recorded in the Danish red data book (Stolze & Pihl 1998) now considered as extinct, endangered or vulnerable (subset = red-data). Red-data were selected from the remaining samples in DANVEG and samples without cryptogams were included, leaving us with 78 samples containing 23 different threatened or extinct vascular plant species.

The last subset consisted of 9 samples from abandoned fields on infertile soils, and was derived from species lists provided by Frederiksborg County (Unpublished). This subset was not included in the paper (Ejrnæs et al. 2002).

A priori classification

Training data for the neural network consisted of samples recognised as either semi-natural vegetation or agricultural vegetation. Class “semi-natural” was assigned to all samples from subset semi-natural. Class “agricultural” was assigned to those samples from the 1998-inventory having calibrated Ellenberg-values (Ellenberg et al. 1992) indicating ecological conditions outside the range observed for class “semi-natural”. This was based on the reasoning that fertilisation, and often also drainage have changed the vegetation of agriculturally improved sites.

Ellenberg-values (Ellenberg et al. 1992) are indicator-values describing the ecological optima of a large number of European plant species along ecological gradients (e.g. water and nutrient availability, temperature and light). Environmental calibration based on such species indicator values are used increasingly for ecological interpretation of floristical gradients when environmental data is scarce or missing (e.g. Ejrnæs & Bruun 2000). Several studies have used environmental calibration for the analysis of vegetation changes associated with human pressures such as atmospheric deposition of acidifying substances and nitrogen (e.g. van Dobben et al. 1999; Lameire et al. 2000).

Calibrations were obtained by averaging Ellenberg-values over all vascular plant species in a sample. In order to select sensible limits for class “agricultural”, Ellenberg-calibrations of all samples were used to delimit a section that contained as few representatives of class “semi-natural” as possible. This section corresponded to samples with calibrated Ellenberg nitrogen > 5.4 and Ellenberg humidity < 7.3. This section contained 3 samples from class “semi-natural” and 334 samples from the 1998-inventory. The 334 1998-samples hereafter comprised class “agricultural”, and judged from the field notes of the inventory they may be interpreted as agriculturally improved dry grasslands and moist meadows and set aside fields. The remaining 631 samples from the 1998-inventory were saved as test data for model evaluation. Table 1 gives a summary of the data sets used in the study.

Table 1. A summary of the five data sets used in the study and their role in the analysis.

Defined subsets	Origin	Role in analysis	# samples
Subset semi-natural	DANVEG	Training	333
Subset agricultural	1998-inventory	Training	334
Test data	1998-inventory	Testing	631
Red-data	DANVEG	Testing	78
Abandoned fields	Unpublished data	Testing	9

Ordination

We used ordination to condense the species matrix into the most important floristical gradients, excluding rare species and species-poor samples prior to ordination (species < 5 occurrences and samples < 5 species). As recommended (e.g. Minchin 1987, Ejrnæs 2000) we applied two different ordination methods – detrended correspondence analysis (DCA, Hill 1979) and non-metric multidimensional scaling (NMS, Kruskal 1964). Ordinations were carried out in PC-ORD (McCune & Mefford 1999). The full data set including training data and test data but excluding red data was subjected to ordination.

Neural network modelling

The classification problem of this study may be seen as a non-linear multiple logistic regression problem. Because we find interactions between our predictors (a gradient in water availability and a gradient in nutrients) very likely, we prefer a classification method that may take into account not only non-linear relationships but also locally changing relationships. A (artificial) neural network (NN) is one possible approach to such a problem (Venables & Ripley 1997).

The neural network used here is a simple, classical NN. It is composed by: 1) a number of input units corresponding to the number of predictor variables, 2) an optional number of hidden units that each receive numbers from all input units and apply a fixed logistic function to these before passing them to 3) an output unit producing the results. The network is tuned by adjusting the weights, which are constants multiplied to the values before they are passed from one unit to another in the network (Venables & Ripley 1997).

A disadvantage of NN compared to conventional statistical models is that the process leading from predictors to results is invisible as it takes place in the hidden layer. For this application the lack of visibility is considered a minor problem. First, we only have 2-3 predictor variables with an expected causal relationship to the response variable. Second, the solution to the classification problem may be visualised and evaluated with a two-dimensional graph. Another claimed disadvantage of NN is the tendency to converge to several locally optimal solutions, depending on the initial weights. Rather than worrying about this, we have chosen to focus on the performance of the best cross-validated solution.

NN does not differ from other modelling approaches with respect to the risk of over-interpretation of data. In NN the degree of fitting may be decreased by reducing the number of hidden units and the variance of the weights (increasing weight decay) (good examples in Ripley 1996). Values for these parameters were selected based on cross-validation (see below).

Cross-validation

Cross-validation (Venables & Ripley 1997) was used to choose sensible values for size and weight decay as well as to select the best ordination method (DCA vs. NMS) and the optimal dimensionality (two vs. three ordination axes).

In order to carry out cross-validation, 10 random subsets were drawn from the training data. For each subset, the model was fit to the remaining 9/10 of data and used to predict the class membership of the left-out subset. Hence, model predictions were evaluated by comparison of predicted class (all subsets) with observed class (the a priori classification). This procedure was repeated 4 times with different random subsets and the mean number of wrong predictions was used as test value for the performance of the specific NN.

Prediction and evaluation

The optimal NN based on the cross-validation was fit to the full training data set. 10 NN-solutions were produced to assess the stability and the best performing solution was accepted as the final NN-classifier. This classifier was used for predicting the value of test data to either class “semi-natural” or class “agricultural”.

Evaluation of the prediction of test data was performed on four selected measures of conservation value: richness of native species, percentage of native species, occurrence of uncommon species and β -diversity (Whittaker, 1972).

Subset “red-data” and subset “abandoned fields” were used for evaluating the performance of the classifier when applied to new data. These samples were not included in the initial ordination and so the DCA-scores were first to be determined by passive ordination (Økland 1990) based on the species scores from the existing DCA solution. The red-listed species were not present in the data used for ordination and therefore had no influence on the calculation of sample scores by passive ordination.

Results

Three ordination solutions were produced – a 3-dimensional DCA, a 3-dimensional NMS, and a 2-dimensional NMS. DCA1 corresponded to a gradient in productivity (nitrogen) and pH from fertilised grass leys to infertile heathlands and oligotrophic mires, and DCA2 corresponded to a gradient in humidity from dry grasslands to wet fens, mires and bogs. NMS1 and NMS2 showed significant correlation with DCA1 and DCA2. Neither NMS3 nor DCA3 could be interpreted in ecological terms.

We used cross-validation to examine the performance of NN's with nine combinations of a range of settings for size of the hidden layer and weight decay. A simple NN, with 2 units and weight decay=0.001 performed equally well as any more complicated NN and was therefore selected for the final model. The misclassification error of this model was only 1.1 %.

Table 2. Comparison of the two classes semi-natural and agricultural derived from prediction of test data. Species richness, nativeness, samples with uncommon species and β -diversity are reported as well as statistical tests for class differences.

Measure	Test	Semi-natural	Agricultural	Probability
Species richness (native species)	Wilcoxon test	16.4	16.9	P>0.05
Percentage of native species	Wilcoxon test	93.9	84.3	P<0.001
Percentage samples with semi-rare species	Chi-square test	38	27	P<0.01
β -diversity in 10 random draws of 40 plots	T-test	12.96	9.42	P<0.001

The cross-validation experiments further revealed that NN performed better on DCA-scores than on NMS-scores. Inclusion of DCA3 in the model did not improve the cross-validated predictions.

Evaluation of test data classification

Figure 1 shows the final NN-classifier. All plots are shown in ordination space with contour lines indicating the probability of membership to class semi-natural (high quality) as predicted by the NN-classifier. The plot confirms our initial expectation of a strong relationship between composition of the vegetation and the a priori classification.

Evaluation of the classification of test data (Table 2) revealed that samples predicted to class semi-natural differed significantly from those going to class agricultural by having a higher percentage of native species, a higher number of plots with semi-rare species and higher β -diversity. The richness of native species did not however differ between the two classes.

The red-data set was subjected to passive ordination using DCA-scores from the ordination. The subsequent classification resulted in all 78 samples containing threatened plant species being classified as semi-natural with a probability > 99% (Figure 2).

The motivation for a succession model

The evaluation convinced us that the classifier was capable of identifying samples of high conservation value, and also that the prediction of samples with low value was trustworthy. It is important however to look also for false-positives, i.e. samples of low value that are predicted to have high value. The habitat quality model use for its prediction vegetation gradients reflecting productivity and hydrology. The most obvious false-positives would therefore be young phases in the secondary succession of vegetation on disturbed infertile soils, e.g. abandoned fields or gravel pits. To evaluate this hypothesis we used species lists from 9 fields on infertile soils abandoned 25-30 years ago. As seen in figure 3, these fields were all predicted to have high habitat quality by the model. Judged from the species lists this was clearly not satisfactory. Although some common grassland species had colonised the field (e.g. *Agrostis capillaris*, *Hieracium pilosella*), the fields were still poor in typical grassland species and contained a large number of species considered alien to semi-natural grasslands on poor sandy soils. This in turn motivated the development of a succession model.

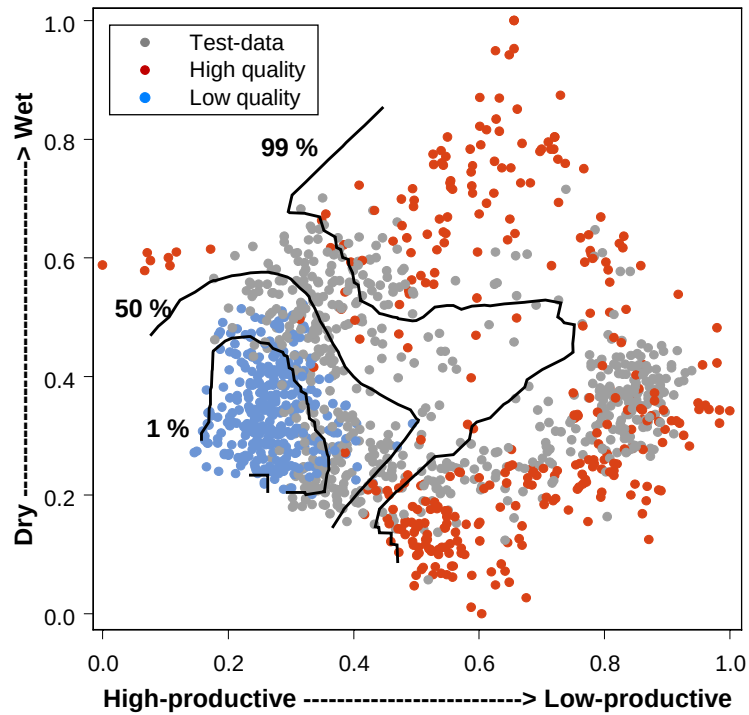


Figure 1. A graph showing the ordination coordinates of training data (high and low quality) and test data. Along the axes are written the environmental interpretation of the gradients in species composition. Contourlines show the three levels of probability of high quality as predicted by the neural network classifier.

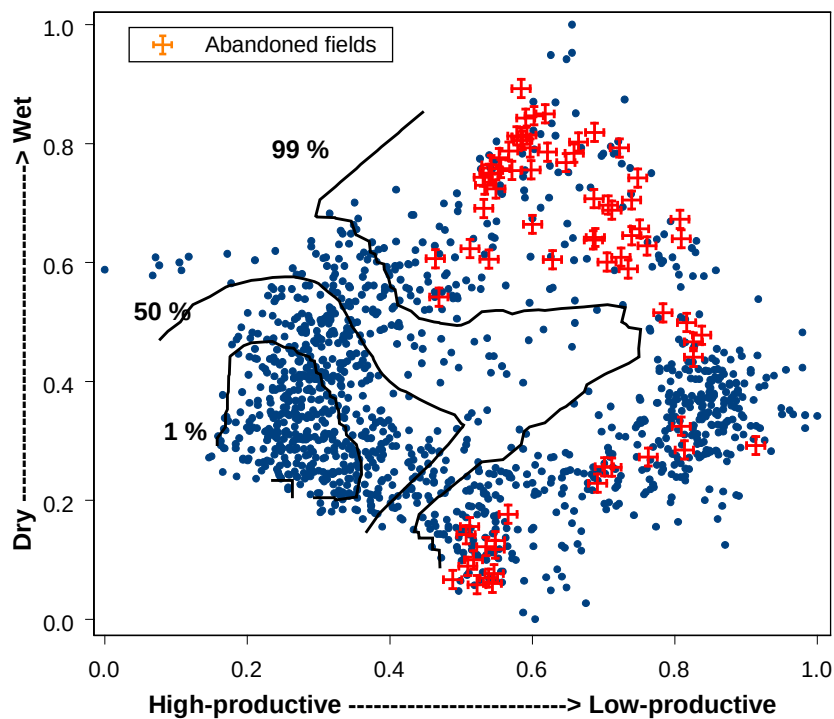


Figure 2. The graph shows the same plots and contourlines as in figure 1, but in addition is showed the position within the reference data ordination of 78 samples with redlisted plant species (positioned by passive ordination).

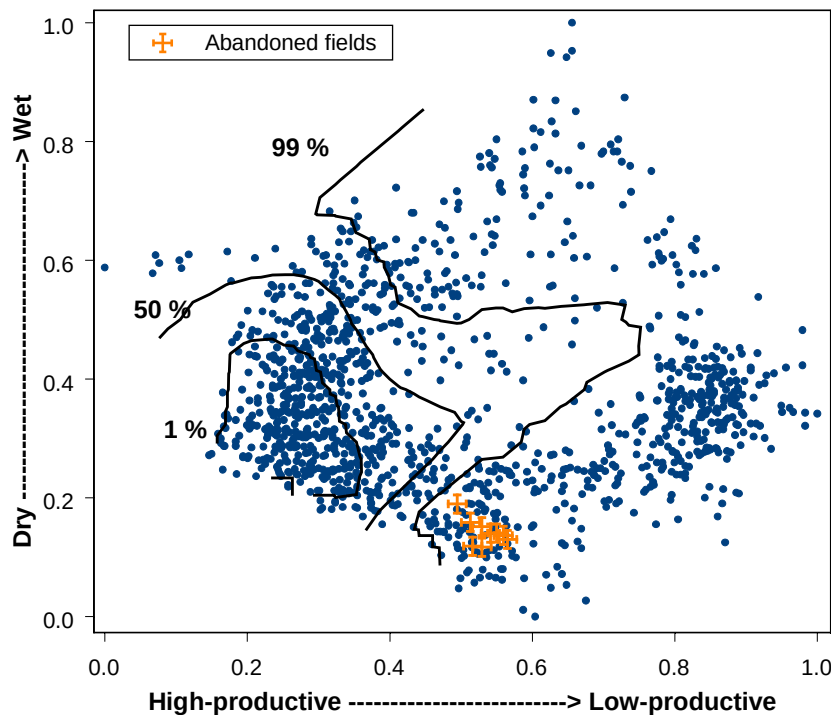


Figure 3. The reference-plot from figure 1 again, but this time with plots from abandoned, infertile fields, placed by passive ordination.

The succession model

For the succession model, we collected a reference data set consisting of published and unpublished Danish studies of grassland, heathland and abandoned field vegetation. Our criteria for including data were: 1) complete species lists from homogeneous areas, 2) species lists classifiable to either grassland or heathland with long continuity, or abandoned fields and 3) if abandoned fields, years since abandonment should be known. We included 13 data sets, with the majority of studies from 1930-1950, and 1970-2000. Data include 620 samples of old grassland, 904 samples of old heathland and 535 samples of abandoned fields. Although abundance data were collected in some of the studies, and some authors recorded lichens and bryophytes, we decided to use only presence-absence data of vascular plants for this study. The sample plots vary considerably in size from 10 m² up to approx. 1 ha. The final data set consisted of 2059 samples and 601 species.

Numerical methods

Data were a priori classified as “abandoned fields”, “old grassland” or “old heathland” and the objective was to predict the a priori classes from information derived from the species lists. We used DCA for ordination. We omitted species with less than three occurrences in the data set prior to ordination but otherwise used default options. As for the Habitat Model, a feed-forward neural network (Venables and Ripley, 1997) was used to find a solution to the classification problem.

After parameter selection (selected after cross-validation), we produced 100 NN-solutions, and accepted the mean probability of the five best solutions as the predicted probability of a sample being typical of a habitat with long continuity.

Evaluation of classifier

Besides using prediction error as a measure of classifier performance, we evaluated the reliability of the NN-classifier of successional phase by establishing and testing the following four hypotheses regarding model predictions:

- 1) For abandoned fields, we hypothesise a significant positive relationship between the predicted probability of being old and time since abandonment.
- 2) Considering that agricultural practices have intensified and colonisation sources for grassland and heathland plants have diminished over the last 60 years, we hypothesise that the probability of reaching a grassland or heathland condition have decreased over time.
- 3) We expect a marked difference in time-span between succession towards heathland and succession towards grassland. A successful colonisation of very few species, mainly heather (*Calluna vulgaris*), may transform an abandoned field into a functional and structural heathland within few years, whereas a grassland succession implies the colonisation of a larger number of typical species.
- 4) A considerable part of the abandoned fields would be classified as semi-natural by the Habitat Quality Model (Ejrnæs et al. 2002) because they are infertile. A major part of these would be correctly recognised as abandoned fields by the new Succession Model.

The first three of these hypotheses were tested using linear modelling of predicted probability of being old as a function of successional age, period of abandonment and successional trajectory (succession towards heathland vs. grassland). The predicted probability was arcsin-square-root transformed to normalise residuals (Zar 1996). To obtain reasonable abandonment periods, we divided data in three groups: Fields abandoned in the period of 1874-1949, 1949-1969 and 1970-1999. In order to classify the successional trajectory we used the NN-classifier. For each abandoned field, the habitat type with the highest predicted probability defined the trajectory. Second order interaction terms were allowed in the model, but the third order interaction term proved insignificant in a trial model and was left out of further consideration. Significance of factors and second order interactions were tested using type III sum of squares (SAS Institute, Inc. 1990).

For testing the fourth expectation, we used the Habitat Quality Model (Ejrnæs et al. 2002) to classify the habitat quality of the abandoned fields and then used the new model to predict the successional phase.

Results

The first three DCA-axes were used as predictors in the classification model. The optimal NN consisted of one hidden layer with two hidden units, and a weight decay of 0.01. Table 3 reports the misclassification rate of the mean prediction of the best five NN.

Table 3. Misclassification rates and misclassification score from neural network classification is given in first column and the corresponding cross-validated misclassifications in second column. Row one to four show misclassification errors between class pairs.

	Misclassifications	Cross-validated misclassifications
Field to grassland	8.4 %	9.1 %
Field to heathland	12.3 %	12.5 %
Grassland to field	3.4 %	3.3 %
Heathland to field	0.4 %	0.7 %
Overall	6.6 %	6.9 %
Misclassification score	187	200.8

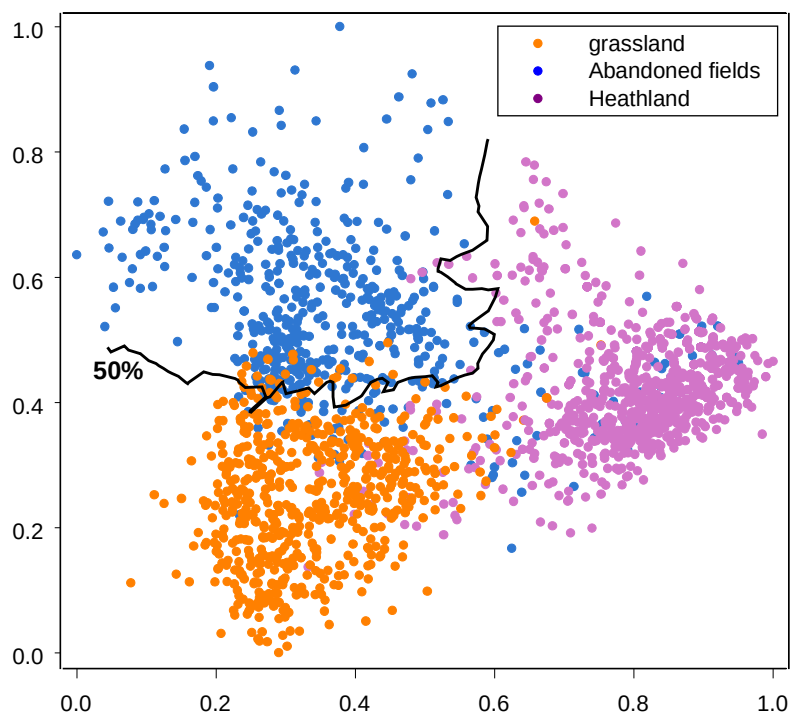


Figure 4. DCA-graph showing the position of the training data set samples of the Succession model and the 50 % contour line of the neural network classifier.

The overall misclassification rate was 6.6 %. This figure hides important differences between the classes however. The percentage of semi-natural plots mis-classified as abandoned fields was generally low. Heathland misclassified as abandoned fields was below 1 % and grasslands between 3 and 4 %. The percentage of abandoned fields misclassified as semi-natural was considerably higher. 12.3 % was classified as heathland and 8.4 % as grassland. The cross-validated predictions were very close to the model predictions indicating a reliable degree of fitting by the NN.

Table 4. Results from a linear regression model of probability of semi-natural condition as a function of successional age, period of abandonment and successional trajectory. The table shows for each variable or interaction term in the model, the degrees of freedom, Type III Sum of Squares, F-value and p-value.

Variable	Df	SS	F-value	P-value
Period	2	6.4	50.7	<0.0001
Trajectory	1	1.6	25.0	<0.0001
Age	1	0.4	6.3	0.013
Period * Trajectory	2	8.4	66.5	<0.0001
Period * Age	1	0.1	0.9	0.42
Trajectory * Age	1	1.0	15.3	0.00011
Residuals	524	33.2		

A discrete clustering in ordination space of our three target classes could cause such a nice discrimination, but this is obviously not the case (fig. 4). In general, plots disperse continuously along the three DCA-axes, although heathland plots, at the end of the first axis, seem to occupy a more isolated part of the ordination space.

Table 4 shows the modelling results, and, in agreement with our hypotheses, successional age, abandonment period and succession trajectory (grassland versus heathland) all have significant impact on the probability of being semi-natural. Also, the interactions of trajectory and period as well as the interaction of trajectory and age are significant predictors in the model. The full model has an r-squared of 0.54 and is highly significant ($p < 0.0001$).

Figure 5 visualises the model for prediction of late successional phase. The graphs demonstrate that:

- 1) The probability of late successional phase increases with age [age significant in Table 4].
 - 2) The probability has decreased dramatically from the beginning to the end of the 20th century, especially for heathland successions [period and period x trajectory significant in Table 4].
 - 3) Heathland successions, especially in the early period, are more likely to reach semi-natural condition than grassland succession [trajectory and trajectory x period significant in Table 4], and typically reach late successional phase much earlier [trajectory x age in Table 4].
- The considerable variation in probability along the regression lines should be emphasised too.

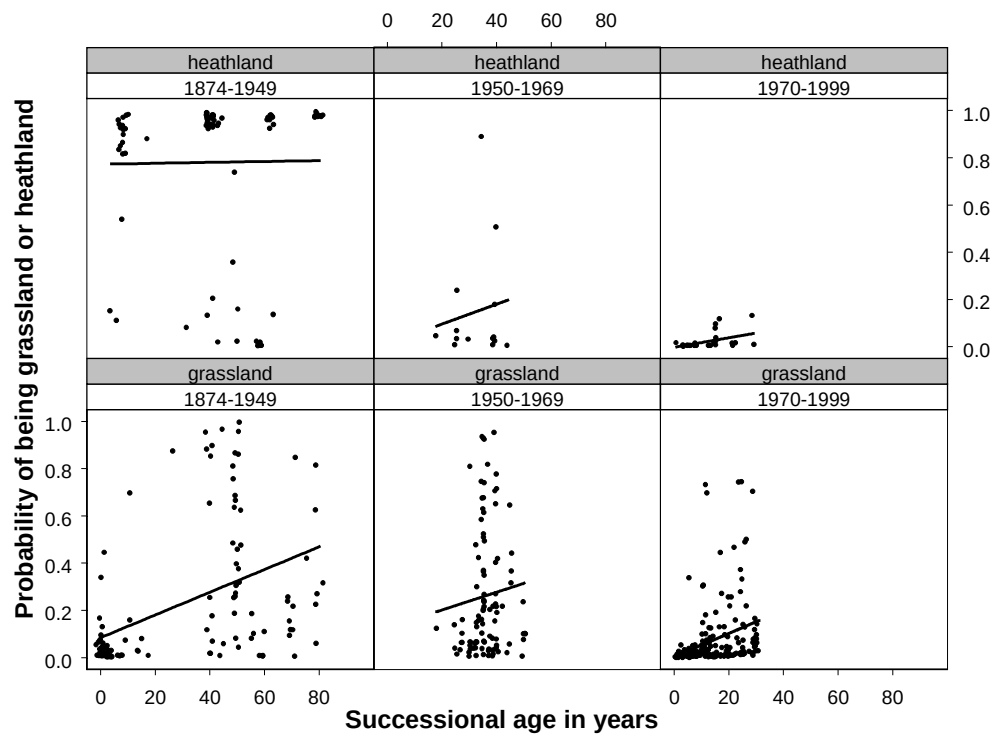


Figure 5. A panel plot of the classification model for probability of late successional phase. Sample plots and contour lines for model probabilities are plotted along DCA-1 and DCA-2 in subplots representing segments along DCA-3. All axes are normalised to the interval 0-1. Abandoned fields are represented by squares, grasslands by circles and heathlands by triangles.

	Late successional	Early successional	Totals
High quality	112	308	420
Low quality	2	113	115
Totals	114	421	535

Table 5. Cross-table comparing predictions from the Habitat Quality Model with predictions from the Succession Model. Chi-squared test value = 31.99 corresponding to a significance level $p < 0.0001$, indicate a significant difference in predictions of the two models.

Table 5 compares the predictions of the Habitat Quality Model (Ejrnæs et al., 2002) with the prediction of the Succession Model. Out of 535 abandoned fields, the majority, namely 420 plots, are predicted to be of high habitat quality by the Habitat Quality Model. The remaining 115 plots are predicted to have low quality. The two models agree when it comes to the low quality plots – 113 out of 115 is also predicted to be in early successional phase. But, as expected, a large fraction of the high-quality plots (308 out of 420) are predicted to be in an early successional phase. The disagreement between the two models is highly significant.

Discussion and perspectives

Perspectives

In this paper we have described methods based on pattern recognition with reference to ordination gradients of a reference data set. The advantages of this approach are multiple. Evaluation of biological condition from species lists can now be carried easily using standardised methods with documented statistical and biological properties. This also means that the principles and criteria for legislative protection and biological evaluations can be presented to the public in the form of standardised methods rather than subjective judgements. The achieved clarification of the legal rights of landowners is considered highly attractive. The classifiers can be implemented on the World Wide Web with free access to end-users such as local and national managers, consultancy companies, landowners and schools.

Unlike subjective evaluations of conservation status, the classifier and its visual component, the ordination graph, may be used also to interpret small directional changes over time. This is demonstrated in figure 6, where a time series covering 26 years of secondary succession in an abandoned field on dry infertile soil (Degn, 2001) is projected on the ordination of the succession graph. The graph visualises a secondary succession that reach the boundary between typical abandoned fields and typical grassland/heathland vegetation after approximately 25 years.

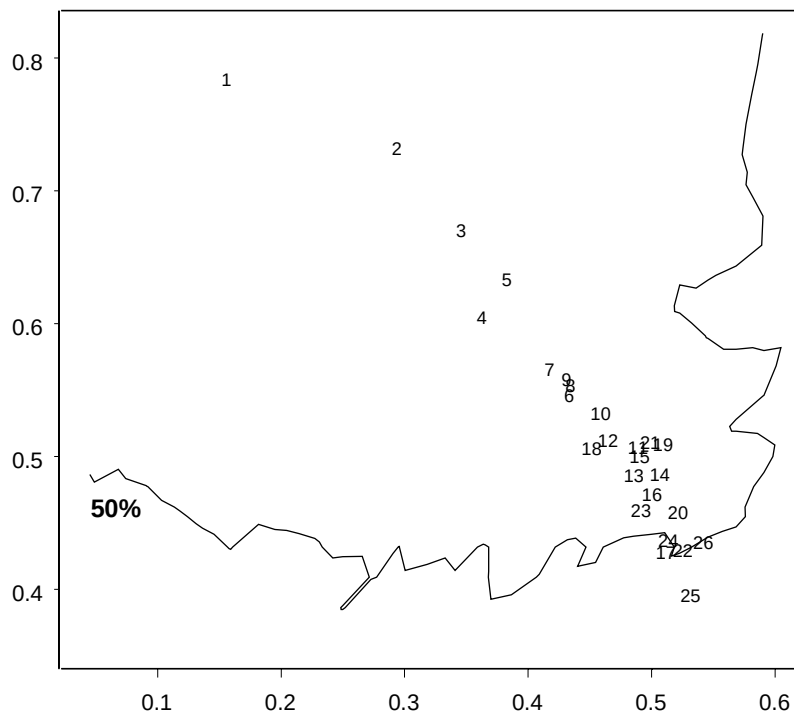


Figure 6. Projection on the succession model ordination of a 26-year time series of secondary succession on an infertile abandoned field (by passive ordination). The 50 % probability line of the classifier is indicated. The graph only shows a section of the graph in figure 4.

This almost unidirectional change from abandoned field to grassland/heathland did not appear as evident from the separate analysis of the time series alone (Degn 2001). We have shown two practical applications of the concept, but more can easily be imagined. Take for instance the definition and delimitation of Annex I types of the Habitats Directive. These are defined in the Interpretation Manual (Anon. 1999), but the definitions may be hard to use in practice, especially in countries marginal to the geographic region covered by the directive. A reference-based classifier based on relevant sample data from the region in question may here be used to specify what the typical habitat looks like, how it may vary naturally and how it is distinguished from other habitat types (Ejrnæs et al. in press).

Limitations

The concept also presents problems and limitations. It may sometimes be hard to collect relevant reference data. Although most European countries have large databases with species accounts suitable for ordination, biologists have had a tendency to collect data in localities of high conservation value and data are therefore often sparse in localities with high human impact. Data may also lack for rare or transitional vegetation types. In some occasions there are plenty of data, but it is questionable whether the sampled species group is appropriate or optimal for indication of conservation value. We have for example in Denmark plenty of data on vascular plant species in forest plots, but here, lichens, bryophytes, beetles and wood decaying fungi are possibly more relevant indicators of conservation value.

Even in situations with large suitable reference data sets, it is worth noticing the limitations of evaluations based on ordination gradients. Ordination is based on the species present in the list, whereas absent species are generally disregarded. As mentioned above ordination does not reflect species richness but species composition. That means that if an environmental change leads local extinction of a number of rare species without any immigration of new species, then ordination may not detect this change as long as the remaining species are characteristic for the reference conditions. Although this may not be a common ecological situation, one may imagine cases where the first subtle response to environmental changes implies a disappearance of a number of particularly vulnerable species, but no immigration of “alien species”, and no major changes in the relative frequency and abundance of surviving species.

The biased focus on species composition is related to the limited sensibility with respect to detection of the first subtle change in species richness. It may be argued that the first change following a human alteration of environmental condition is to be detected in changed environmental conditions rather than altered species composition (delayed response). It may therefore be argued that evaluations based on species composition ought to be combined with evaluation of monitoring data on both the population trends of vulnerable species and on the physical and chemical condition of the habitats.

Scientific consensus about reference condition

As a final remark, we would like to stress the importance of scientific agreement about the concept of reference condition (also see van Hinsberg et al. 2003). Such consensus is paramount and a prerequisite for establishment of reference data sets for evaluation of states and condition in biodiversity. We advocate for two levels of reference condition.

The first level is essentially the condition that would prevail if no human alteration of the environment had taken place – i.e. the natural baseline of species, habitats and ecosystems. This baseline is essential to our scientific understanding of the ecological and evolutionary preconditions for biological diversity. We suggest calling this the *natural baseline*. The natural baseline should be defined with reference to the state of the art knowledge of palaeoecology and evolutionary ecology. And, acknowledging the limited biotic evidence available from disciplines such as palynology, we will need to complement this knowledge with insights from

community ecology and autecology. It is inevitable that the statement of natural baseline will include elements of uncertainty.

The second level is a baseline that defines the degree of naturalness that is achievable and affordable given the political and ecological reality of today's world. We may call this a *pragmatic baseline*. The pragmatic baselines should refer to the natural baseline, but need to take into account the changes that humans have enforced on landscapes, habitats and species pools, some of which are irreversible, e.g. global extinctions of keystone species and conversion of natural forest to arable land. The pragmatic baselines should be applicable as objectives for the management. If we take a habitat type as example, the pragmatic baseline should specify the acceptable species composition, species richness, ecosystem properties (nutrient pools and cycling, natural disturbance regime etc.) as well as its area and geographical distribution. Actually this is precisely what the Habitats Directive asks for. The pragmatic baseline will constitute the objective against which biological condition as reflected by monitoring data and effects of remedial management will be measured. The pragmatic baseline defines a target for nature management, whereas the time-span needed to reach this target will depend on the willingness of society to invest the required resources.

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Informing Policy-makers about changes in Biodiversity

A Dutch approach to the aggregation and processing of monitoring data for the achievement of national Biodiversity indicators like the Natural Capital Index

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Introduction

Human influences have caused a rapid acceleration in the rate of biodiversity loss throughout the industrial era due to human influences. According to the Global Biodiversity Assessment (UNEP, 1995), species are now becoming extinct at 1,000-10,000 times the natural rate. However, extinction is the final step in a long process of ecosystem degradation, in which a decline in the abundance and distribution of many species is usually accompanied by an increase in the abundance of a few other species. Generally some common species are becoming commoner, while rare species are becoming more rare.

The Convention on Biological Diversity (CBD) requires participating parties to report on the progress they have made in restoring and maintaining biodiversity and henceforth to implement monitoring. The EU-Habitat Directive states that countries should also set up a system for exercising surveillance of the conservation status of particular natural habitats and their species. Lack of knowledge on the consequences of policy measures on biodiversity hampers optimisation of the planning cycle. Biological diversity is monitored to some extent in many European countries. However, the use of monitoring results in decision-making is often obstructed by poor communication between ecologists and decision-makers. In practice, not all measurements in the field are adequate for providing the necessary information for policy making. An important limitation in this respect is the gap between monitoring programmes and biodiversity indicators needed for communicating with policy-makers. National reports on the implementation of the CBD show that only 6% of the Parties to the CBD have identified national indicators of biodiversity.

In this paper we will describe the Natural Capital Index framework (NCI), an indicator designed for informing policy-makers on biodiversity and developed as a contribution to the implementation of the CBD (UNEP, 1997a). We will also describe the monitoring schemes used for calculating the Dutch NCI and give examples of its application of the NCI in the national Nature Outlook 2 (RIVM, 2002). Finally, we will discuss the applicability of the NCI in other countries and introduce alternative ways for calculating it depending on availability of monitoring data.

The Natural Capital Index framework: the theory

The loss of biodiversity has two main components: i) *loss of habitats*, or “ecosystem quantity”, and ii) *loss of ecosystem quality*. Figure 1 provides a simple visualisation of this process (Ten Brink, 2000; Ten Brink and Tekelenburg, 2002). The grey cut-outs illustrate the loss of habitat area, while, in the remaining areas, the decline in ecosystem quality is shown by the decreasing abundance of characteristic species. During this process of biodiversity loss, species richness is initially often increased by the introduction of exotic species.

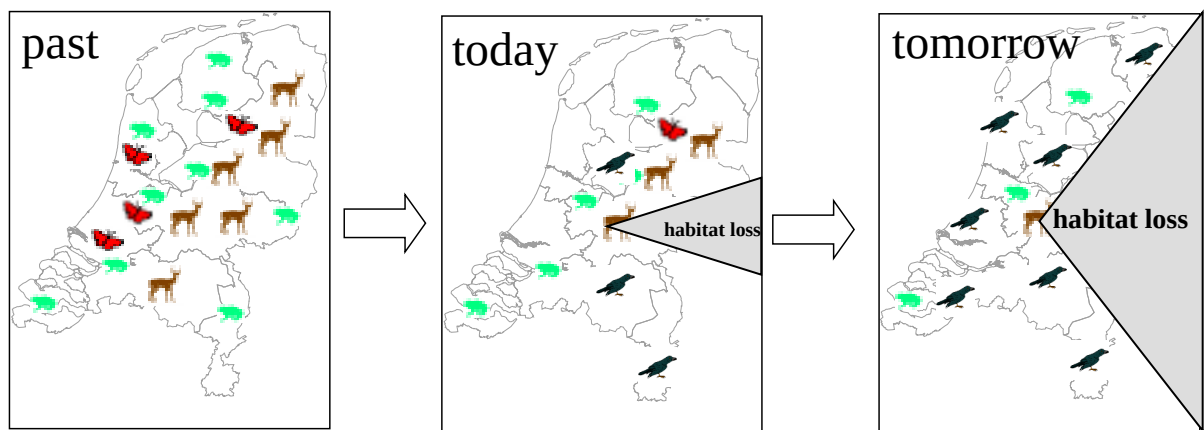


Figure 1. Schematic visualisation of the process of biodiversity loss.

The challenge is to create a tangible, powerful composite indicator that accurately describes the above-mentioned process for meeting policy requirements. Such an indicator must also be appealing to policy development, quantitative, sensitive, affordable and, measurable. Finally, the indicator must be linkable with socio-economic scenarios.

The Dutch Office for Environmental Assessment, which has the legal task of informing Dutch policy-makers on environmental issues, uses two types of indicator for this purpose: i) the Species Group Trend Index (STI) and ii) the Natural Capital Index (NCI). The STI focuses on changes in abundance of particular groups of species, like breeding birds in the open dunes. The NCI focuses on natural capital as the product ecosystem quantity and ecosystem quality (Figure 2):

$$\text{NCI} = \text{ecosystem quantity} * \text{ecosystem quality}$$

Ecosystem quantity is defined as the size of the ecosystem, expressed as the percentage area of a country or region. Ecosystem quality is defined as the percentage of the quality relative to a baseline situation. The quality is based on information of abundances of a core set of animal and plant species. However, other variables too, like ecosystem process and structures might be used as quality variables. The baseline represents the natural or low-impacted ecosystem.

The ecosystem quality, quantity and the resulting NCI range from 0 to 100%. A quality of 100% signifies an abundance of all species equal to the baseline conditions. A quality of less than 100% signifies an abundance in which one or more species has decreased below baseline conditions. The changes in biodiversity, as depicted in Figure 1, can be visualised using the NCI (see Figure 2).

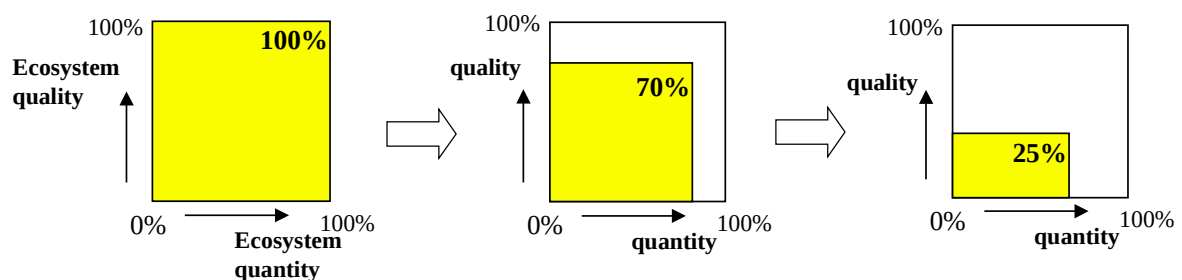


Figure 2. NCI used to visualise the process of biodiversity loss depicted in Figure 1.

A baseline is essential to determine ecosystem quality. Without a predefined and explicit baseline it would be impossible to tell policy-makers whether ecosystem change is good or bad. Baselines are common practice in medical care, economic development and climate change. In ecology, pre-defined baselines are not yet common practice. The first CBD Liaison Group on Biodiversity Indicators recommended to use “a postulated baseline, set in pre-industrial times” or a “low-impact baseline”, in order to indicate the losses in recent times due to large scale human activities. This historical baseline is not linked with a certain year but with the socio-economic development stage of a country or area. This baseline i) allows interpretation of changes in biodiversity (the impact of modern human activities on ecosystems); ii) makes data within and between regions and countries comparable; and iii) is relevant for all "nature types". In the Netherlands the baseline was a pragmatic description of the "low impact" or "natural" baseline, set to the situation in the 1950's. Where the 1950's were not suitable to describe a low impact state, other historical references, geographical references, or potential states were used to describe the baseline.

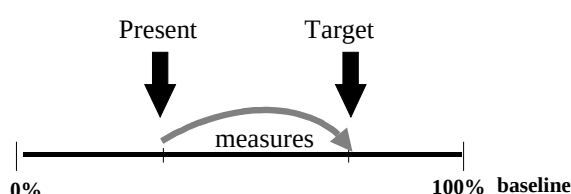


Figure 3. Ecosystem quality is calculated as a percentage of the baseline state.

It should be stressed that the baseline is *not* the targeted state. Policy-makers set the targets. However, policy-makers can use the baselines to set the ecological targets on the axis somewhere between 0 and 100%, depending on the political balance between social, economic and ecological interests (Figure 3). On the other hand, the baseline can also be used to evaluate the ambitions of the policy-makers.

The methods for calculation

Since the NCI focuses on ecosystem quality and ecosystem quality, the first step is the distinction in major nature types. In applying the NCI in the Netherlands we use a set of 32 different nature types, combinations of major habitat types and geographical regions. The next step is to define the set of characteristic species by which ecosystem quality can be measured. For the Netherlands the nature types were described with 980 species. The species were selected from the taxa of vascular plants, birds, mammals, butterflies, reptiles, fish, macrofauna and molluscs. The selection for these groups was mostly driven by the availability of data on species abundance from existing monitoring schemes. The selection of nature types and the species was based on the following set of criteria:

- Nature types and species need to be described using sufficient information, especially with respect to the baseline state
- Nature types and species must be ecosystem and policy-relevant
- Nature types and species must be unambiguously and affordably measurable
- Changes in the nature types and species must be suitable for modelling
- Nature types must cover all Dutch ecosystems, whereas species must be characteristic and representative as a whole for a particular nature type
- Nature types and species must be able to show the influences of human pressures
- Nature types and species must be sensitive enough to detect changes, but should not be over- sensitive

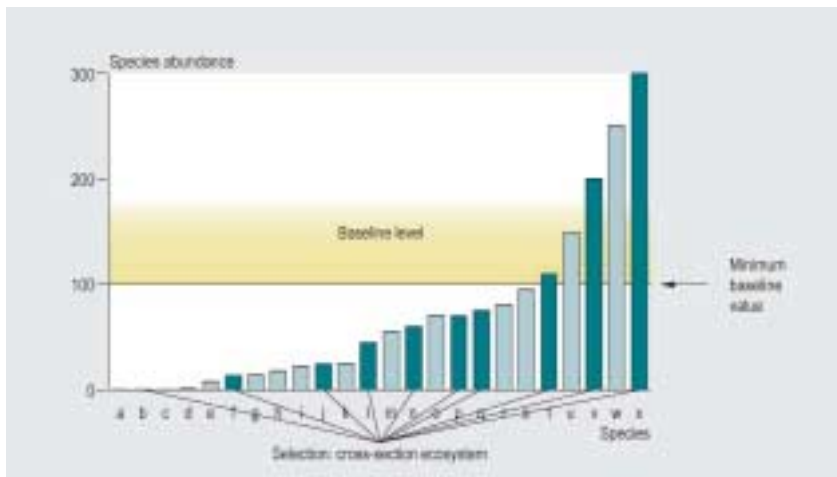


Figure 4. Smart sampling must be used to describe ecosystem quality.

The selection of species is the result of an iterative process, considering the species criteria of the indicators and the (cost-effective) possibilities of the monitoring schemes.

Since it is neither necessary nor possible to include all species in a biodiversity indicator, a representative cross-section of characteristic species must be used to describe the process of biodiversity loss. Focusing only on the very rare species will give a sensitive indicator for detecting changes when quality is high. However, in detecting improvement after policy-measures have been taken, the focus on rare species will not deliver a sensitive indicator nor will it provide a representative picture of the entire ecosystem. Smart sampling can be used to create a sensitive and un-biased indicator (Figure 4).

Calculation principles

The last step in the calculation of the NCI is to define the calculation principles, which describes how the NCI can be determined from monitoring schemes and ecological models. The following procedure is used for calculating the NCI in the Netherlands:

1. Retrieve information about species abundance for each measured or modelled location and calculate the quality index per species per nature type. The quality index is the current value divided by the baseline value, resulting in a ratio in the range of 0 to 100%. The absolute abundance in the baseline situation varies per species but is set at 100%. Abundance can be measured in different units (e.g. chance of occurrence, density or population size). Data on abundance can be retrieved from monitoring schemes or ecological models.
2. Calculate the mean index for each of the different groups of species (in the Dutch case higher plants, invertebrates and vertebrates) as the arithmetic mean of the quality indices of the individual species within the different nature types.
3. Calculate the quality of the different nature types as the arithmetic mean of the mean indices for the different groups of species within the different types.
4. Calculate the total area (quantity) of the different nature types, as percentage of the entire country are, from maps made by means of ground surveys, satellite information, aerial photos or a combination of these.
5. Calculate the NCI per nature type by multiplying quality by quantity expressed as a percentage ranging from 0 to 100; don't multiply quality by quantity if you want the quality and quantity values to be viewed separately. Calculate the NCI per nature type, geographical region, or for the country as a whole. If quality and quantity are kept separate, qualities of aggregations can be calculated through area-weighted averaging.

Calculation of the Dutch NCI using monitoring data

Most of the ecological monitoring in the Netherlands is organised in the so-called Network Ecological Monitoring (NEM). NEM was set up to provide relevant information for environmental and nature policy decision-making on national scale. In 2002 this network comprised 8 species groups of flora and fauna in 20 robust ecological monitoring schemes. Both terrestrial and aquatic systems are monitored, although the network is less developed for the latter. All of these schemes are species-oriented. Vascular plants, birds, mammals, butterflies, dragonflies, reptiles, amphibians, lichens and fungi are monitored in the terrestrial ecosystem. The monitoring schemes are organised around the different groups of species, an approach originating in the work done with non governmental organisations (NGOs), considering their usual focus of interest and knowledge on a particular group of species. If possible, all species within a group are monitored. However, for difficult and large groups only a selection is monitored. Currently, the information of nearly 1000 species is used to calculate the different biodiversity indicators. Most of the budget (ca. 80%), which came to a total 1.4 million euro in 2002 goes to flora and breeding birds. This reflects the high policy- and ecosystem relevancy of these groups, especially with regard to birds, very popular among the volunteers.

NEM is a cooperation of the NGOs, Statistics Netherlands, the Office for Environmental Assessment and, the national and provincial authorities (Ministry of Agriculture, Nature and Fisheries, Ministry of Housing, Spatial Planning and the Environment, Institute for Inland Water Management and Waste Water Treatment and the 12 provinces). The partners have prevented overlapping in monitoring schemes by formulating joint monitoring objectives. Now, one monitoring system has been set up with support of all partners. The partners finance the monitoring. The budget is completely allocated to the NGOs and provinces involved in the data collection and the coordination of the monitoring schemes. This includes the coordination of the fieldwork, education and motivation of the volunteers, and the delivery of results to Statistics Netherlands. The NGOs work primarily with volunteers who are centrally co-ordinated. Altogether about 10,000 volunteers are involved, organised in 15 NGOs. The majority of the monitoring is carried out by volunteers; however, the monitoring of plant species is done mainly by professionals.

Statistics Netherlands plays a central role in the processing of data, collecting the raw data, performing quality checks and making statistical trend analyses. The indices of changes in occurrences of species are calculated with the statistical program called TRIM: TRends and Indices on Monitoring (Pannekoek & van Strien, 2001). This program is designed to deal with data from ecological monitoring schemes, which are often studded with missing values. Statistics Netherlands reports yearly on the quality of each of the 12 monitoring schemes in the Network Ecological Monitoring. These reports are used by the NEM partners to initiate improvements in the schemes. The clear objectives set for each of the monitoring schemes have been very helpful in evaluating the progress in the development of the schemes and thus in stimulating further improvement over the years. On the basis of information of Statistics Netherlands the Office for Environmental Assessment calculates the NCI, which is presented to policy-makers of the national and provincial authorities in the annual Nature Balances and Nature Outlooks. The independent position of Statistics Netherlands and the Office of Environmental Assessment allows the results to be broadly acknowledged.

Examples of how the NCI is used

The NCI was used to assess the current state of nature in the national Nature Outlook 2 (RIVM, 2002). Although the figures here may give an impression of the most frequently used presentation formats of NCI, although other presentations are also possible.

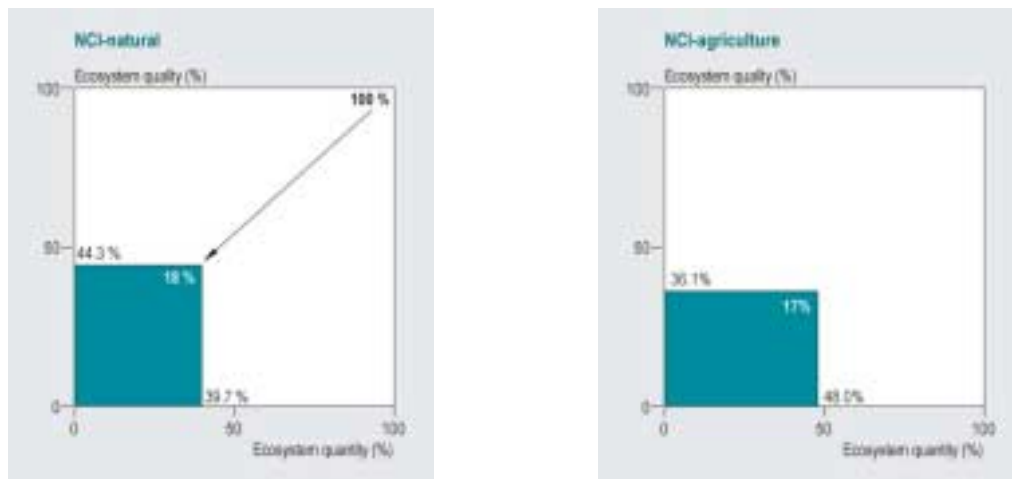


Figure 4. The NCI for natural ecosystems (a) and agricultural ecosystems (b) in the Netherlands.

National policy-makers need information on the current state of ecosystems on a highly aggregated level. Figure 4a shows a decline in the total quantity of natural aquatic and terrestrial ecosystems in the Netherlands of 40% of its total territory, while the average quality of these ecosystems is estimated at a modest 44%. The resulting product of quantity and quality is 18%. Roughly speaking, the average population size of the characteristic species has decreased to 18% in comparison with the baseline. The NCI for agricultural land is 17% (Figure 4b). Although the indices are similar, the NCI for natural ecosystems is, in comparison with the NCI of agricultural land, derived from a smaller area of higher quality.

The data aggregation procedure makes it also possible to calculate the contribution of the various nature types to the Dutch NCI. Figure 5 shows quantity and quality for each nature type. The size of marine and large freshwater ecosystems is very important in the Netherlands, since these ecosystems cover more than 75% of the remaining natural area. The quality of international important nature types like the Wadden Sea, the marshes and the dunes is relatively high compared to the other nature types. Forests, heathland, rivers and inland lakes have the lowest quality.

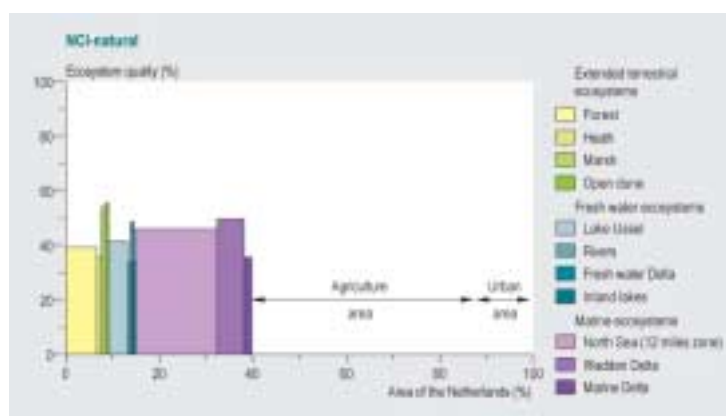


Figure 5. The share of the different ecosystems in the NCI.

Information from figure 5 can also be used to map the current quality of the nature types (Figure 6). The extent of the natural area is determined from land-use maps (resolution of 250 m x 250 m). Ecosystem quality is calculated on a scale for nature types. Terrestrial natural ecosystems in the Netherlands are scattered over the country: estuarine, coastal and aquatic

ecosystems dominating in the west, forest and heathland on sandy soils in the east. All natural ecosystems have a mediocre quality and are seriously affected by various pressures, which is to be expected in this highly populated and industrialised country. The quality of the marine ecosystems and the coastline is relatively high, while that of the inland aquatic systems and heathland is lower.

Figure 6. Map showing the ecosystem quality in the Netherlands on basis of monitoring data.

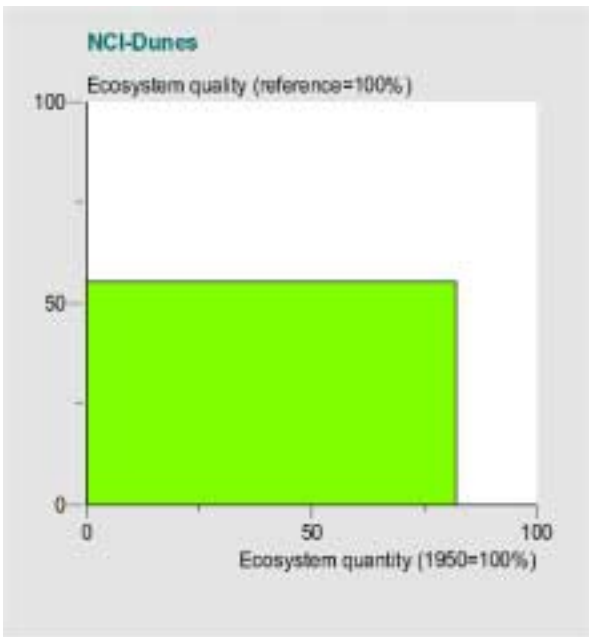
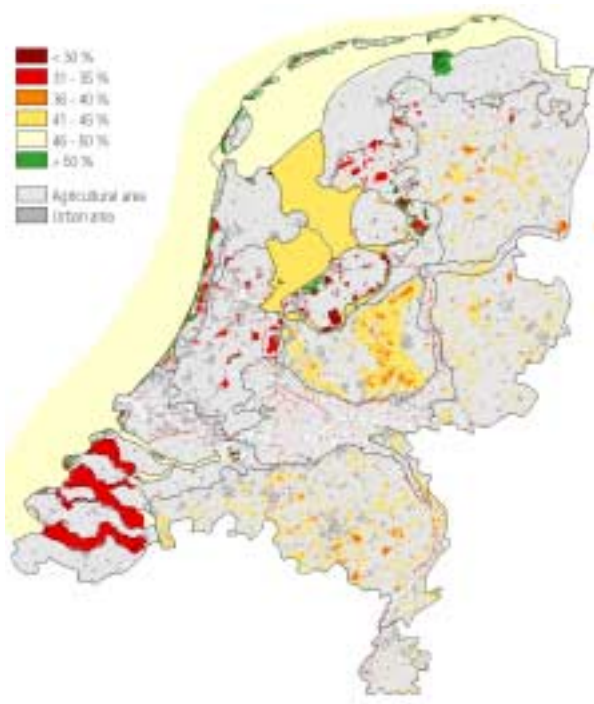


Figure 7. NCI for Dutch dunes. In this presentation the area in 1950 is defined as 100%.

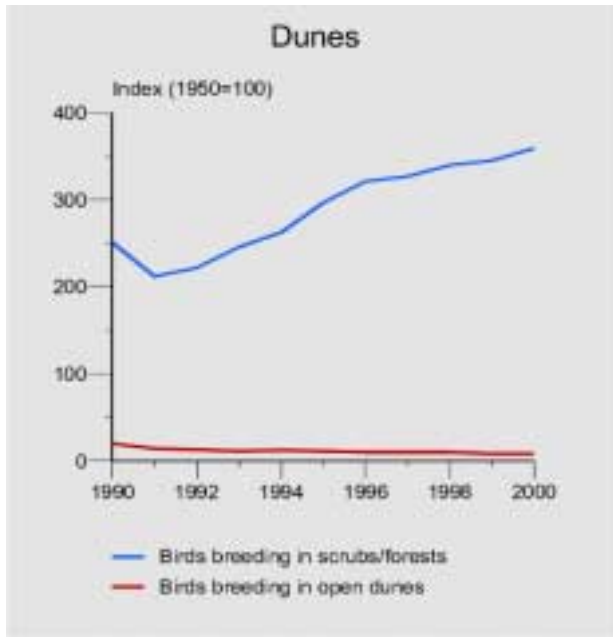


Figure 8. Species Trend Index (STI) for groups of birds in dune ecosystems.

Policy-makers are often also interested in more specific information on the current status of a single nature type, for example, of nature types mentioned in the EU-Habitat Directive. Figure 7 presents the ecosystem quality and quantity of the Dutch open dunes. Even in this relatively well protected ecosystem, both quantity and quality have decreased substantially. As the Outlook focused on status rather than trends, no trends in NCI were calculated. Trends in NCI are scheduled to be reported in forthcoming Nature Balances.

In some cases it might even be useful to focus on detailed information on biodiversity changes within a single nature type. The STI makes it possible to draw attention to a group of species of special interest. The STI can focus on species from the Red Lists, the Birds Directive, the Habitat Directive or the Bonn Convention. The STI can also help in determining the causes of the changes in biodiversity. Figure 8 shows the STI of breeding birds in dune scrubland (Nightingale and Whitethroat) and open dune vegetation (Wheatear, Curlew, Skylark, Redshank and Black-tailed Godwit). In 1990, there were significantly fewer breeding birds in the dunes than in 1950. The species associated with open dunes were scarcer in 1990 than in 1950. These species have been steadily declining since 1990 as a result of the intrusion of scrubland and grasses. The Wheatear has been decimated in recent years. At the same time, a number of birds that breed on scrubland have increased sharply. The intrusion of scrubland is thought to be mainly a consequence of the increase in the supply of nutrients via atmospheric deposition.

By informing policy-makers about the current state of biodiversity, important questions like "What were the main causes", "What will happen in the future", "What are potentials, and what can be done to restore biodiversity in an efficient manner?" have not yet been answered. This is why the Natural Capital Index is linked with ecological models. Figure 9 shows NCI both for the current state and for four different socio-economic scenarios. Here, projections for 2030 have been calculated with ecological models on the basis of policy scenarios, reflecting a range of environmental concerns, societal trends and policy options. The height of the bars indicates the NCI for the country as a whole. The different colours indicate the shares per nature type.

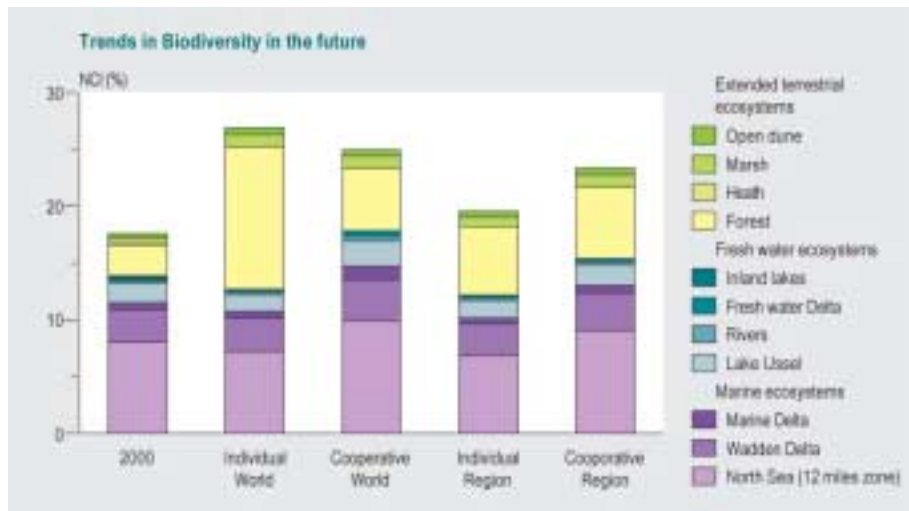


Figure 9. Changes in the NCI by different socio-economic scenarios.

Conclusions and discussion

The NCI concept seems an effective way to communicate with policy-makers about the state and trends in biodiversity. The NCI has become a standard indicator for the Dutch Office of Environmental Assessment. Working with an integrated framework of biodiversity indicators, monitoring schemes and ecological models allows us to answer questions about the current state of nature, possible future trends and the causes of losses.

Although the Dutch NCI can be calculated with the existing monitoring schemes, the design and strategy of the Dutch monitoring schemes have their pros and cons. The clear-cut monitoring objectives related to the delivery of biodiversity indicators provide a clear focus of all parties involved. As stated above, overlapping monitoring schemes are avoided by formulating joint monitoring objectives. However, the consequence of involving all relevant partners, on both supply and demand sides, is a complex organisation, in which decisions on and implementation of optimisations and adaptations go fairly slow. Working with volunteers makes it more difficult to change the monitoring programs. However, thanks to the volunteers a high level of ecological monitoring is maintained despite the current low budgets. The cooperation with Statistics Netherlands also makes an independent check on reliability of the supplied data possible.

Applicability

The use of NCI as an indicator for evaluating CBD objectives will be further discussed in forthcoming sessions of the CBD expert groups on biodiversity indicators. In a GEF-financed project, co-ordinated by the UNEP-World Conservation Monitoring Centre, the NCI-framework will be tested in four different countries (Ecuador, Kenya, Philippines and Ukraine). The use of a standardised baseline as common denominator makes it possible to compare changes in biodiversity both between and within countries. However, the elaboration is country-specific. Depending on the budgets available, the NCI can be implemented in a fairly simple and affordable way, which is important in developing countries with even more limited budgets and availability of data. If data on species abundance is insufficiently available, we propose the use of an alternative calculation procedure in which ecosystem quality is based on information on ecosystem structure or environmental pressures. Environmental pressures can be used as a substitute for ecosystem quality, assuming that pressures are inversely related to quality (Figure 10). Data on pressures are often more widely available. Pressures could be climate change, eutrophication, acidification, fragmentation, etc. Often information is available on current and future pressures based on monitoring and modelling of socio-economic scenarios. If this is the

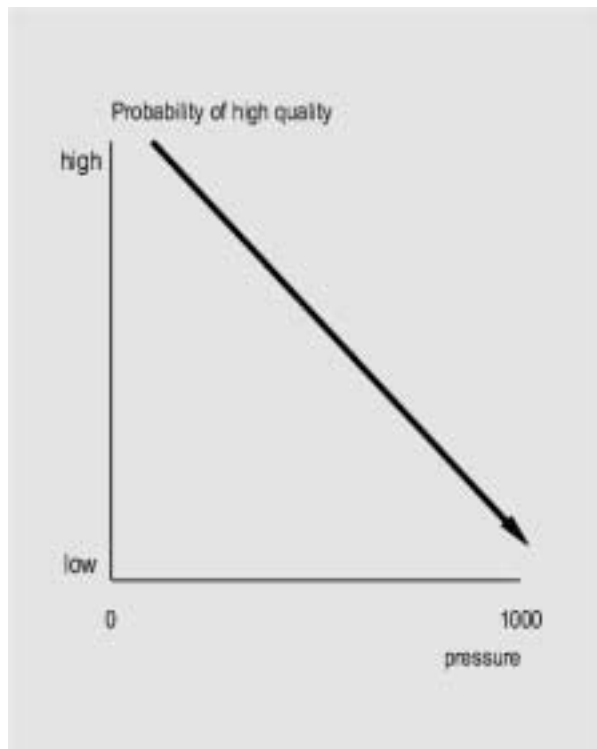


Figure 10. Hypothetical relationship between pressure and ecosystem quality

case, each form of pressure can be graded on a linear scale from 0 (no pressure) to 1000 (very high pressure). A pressure of 1000 means a high probability of extremely poor biodiversity compared with the baseline. The pressure values considered for each area are added to one single Pressure Index, providing an estimation of biodiversity quality. This principle has been applied in GEO 1, 2 and 3 and EU (UNEP, 1997c, 1999, 2002; Ten Brink, 2000; Ten Brink et al., 2001).

We see good opportunities for using the NCI framework in the EU Water Frame Directive, since this is also a reference-based approach describing the ecosystem quality with species and/or species abundance. The baseline in the NCI shows good correspondence with a "good ecological status", defined in the Directive as "representing only slight deviation in the biological quality elements from an *undisturbed* site". The NCI might also be used to communicate with policy-makers about whether the "favourable conservation status" of the EU Habitat Directive is met. However, the Habitat Directive will also give an extra stimulus for collecting distribution data of the priority species. Distribution data are collected for different purposes than monitoring. Distribution information is essential for spatial planning decisions on the local/regional scale, whereas monitoring data is more suitable for evaluating national policy on a regular basis.

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The national biodiversity mapping programme in Norway

- methods and status

Terje Klokke & Arild Lindgaard, Directorate for Nature Management, Norway

Abstract

A national biodiversity-mapping programme is being undertaken in Norway. The municipalities are responsible for the actual mapping, and they receive support and advice from the various county governor officers. The Directorate for Nature Management has prepared manuals to guide the mapping. Each local authority can apply for a one-off government grant. Information generated by this mapping is intended to be used in the local authorities' own land-use management, but will also be fed into national geographical databases. A system for determining the value of biodiversity has been drawn up so that each local authority will be able to prepare a map showing its most important areas for biodiversity. This valuation is described in DN Manual 13-1999, and is briefly outlined here. It is based on four manuals that give guidelines on the mapping of, respectively, nature types, habitats of wildlife species, freshwater localities, and marine habitats. In addition, directions are given on how to merge the categories in the Norwegian Red Data List of endangered species into two divisions. Instructions on how to allocate localities for biodiversity to one of two categories, A- areas (extremely important) and B areas (important) are also given. The programme started in 1999 and a national political target requires every municipality in Norway to map the biodiversity within their boundaries by the end of 2003. This mapping programme is a follow-up of the Rio Convention on Biological Diversity. A total of 320 municipalities have by August 2002 begun their mapping effort, and approx. 70 has produced digital biodiversity maps.

Biodiversity monitoring in Estonia

- from specific case to some general ideas

Martin Zobel, Raivo Mänd, Kristjan Zobel (all three from Tartu University) and Tiit Teder (Estonian Agricultural University)

Abstract

There is a functioning biodiversity monitoring system in Estonia. It was created mostly due to the initiatives of scientists, involved in the study of endangered taxa or communities. Because of that, the methodology is, in most of cases, quite well elaborated. Distribution and dynamics of rare species populations, as well as the status of rare or in some or other sense important plant communities, is monitored. At the same time, the biodiversity monitoring system still lacks generality, and the scientific information collected is not always thoroughly analyzed. We shall propose a more simple and general scheme for biodiversity monitoring, which could be part of nature conservation on the state level.

Mapping of protected habitat types of the Nature Conservation Act in Finland

Pilvi Pääkkönen, Nature Division, Finnish Environment Institute (SYKE)

Abstract

The Finnish Nature Conservation Act lists nine protected habitat types which are threatened and have great ecological value: woods rich in broad-leaved deciduous species, hazel woods, black alder swamp forests, sandy shores, coastal meadows, treeless or sparsely wooded dunes, juniper meadows, wooded meadows and prominent single trees or groups of trees in an open landscape. The objective of the Act is to preserve the ecological value of these habitats. The mapping of the habitats was started in 1998. The aim is to investigate the present state of these habitats in Finland and to gather the necessary data for making conservation decisions.

The sites to be surveyed are selected on the basis of earlier studies and planning documents and by using the knowledge of local experts. With field inventories we examine whether a location meets the criteria of the protected habitat or not. Data is gathered on the vegetation, forest structure, flora and fauna, the representativeness and state of the site, the need for management etc. All land owners are informed prior to the field surveys.

Natural habitats have so far been surveyed in all Regional Environment Centres excluding the Kainuu and the North Savo. The number of locations surveyed to date is approximately 1850, of which just under half, 784 locations, have fulfilled the criteria of legally protected habitats. The mapping has so far concentrated largely on forest habitats and the focus will be shifting to the shore habitats in 2004. On the average, the forest habitats are small, ranging from half a hectare up to several hectares. The main tasks of SYKE in the project have been to define the exact criteria of the habitats and their naturalness, to design the mapping methods and the database, to publish a guidance on mapping of the habitats, to coordinate the mapping and the gathering of the data and to make summaries and analyzes of the results. The mapping data constitutes the baseline for the future monitoring of these habitats and it is stored in the national database. The Regional Environment Centres have to carry out the field surveys and later set the boundaries of the habitats to be protected. Until now 160 such conservation decisions have been made, covering all together 315 ha. Most of the decisions have been made in Southern Finland and the majority of them are of forested habitats.

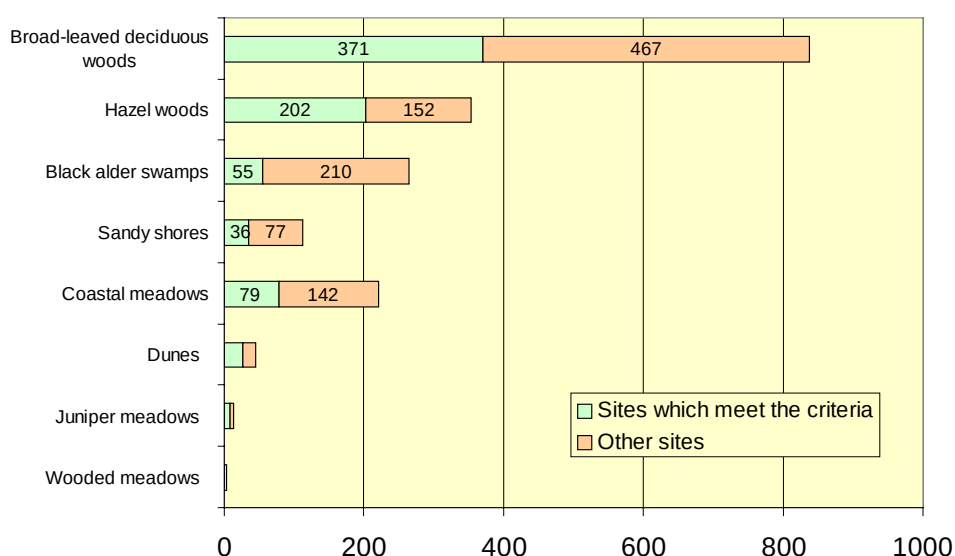


Figure 1. Number of the mapped habitat sites. Situation 1.11.2002.

Developing remote sensing for monitoring of selected habitats of the Habitats Directive.

Johan Abenius, Naturvårdsverket & Biometria, Sverige

Abstract

We describe a number of ongoing projects aiming at habitat monitoring for conservation by use of satellite data. The projects have been carried out by Metria Miljöanalys in co-operation with several institutions including the Swedish Environmental Protection Agency, The Swedish National Space Board, the councils of Kalmar & Norbotten, and has been partly funded by an EC-Life fund.

The purpose of the projects is to support the national and regional environmental monitoring systems with satellite-based methods for describing the state and changes of valuable habitats, primarily for a few selected Natura 2000 habitat types.

The selected habitat types for mapping of current state and changes over time include:

- Western taiga (9010)
- Broadleaved forest types (9xxx)
- Open habitat types (1630, 5130, 6110, 6210, 6280, 6410)

In addition change over time has been mapped for:

Aapamires (7310)

Satellite monitoring of Estonian Landscapes

Kiira Aaviksoo, Estonian Environment Information Center, Estonia.

Abstract

In the framework of Estonian State Environmental Monitoring Programme (EMP) the subproject "Remote Sensing of Landscapes" has been initiated in 1996. Monitoring landscapes and their ecological changes by means of satellite remote sensing has been developed in seven permanent monitoring areas: Alam-Pedja, Endla, Saarejärve, Lahemaa, Soomaa, Karula and Vilsandi. These areas are located in natural landscapes of mire, foodplain, forest and coastal ecosystems. Each monitoring area has a protected area (nature reserve or national park) in its core, and is surrounded by 3 km buffer zone.

Landsat TM data from 1980s to 1990s have been used for mapping and monitoring purposes. The latest image with better interpretation aid - auxiliary material (large-scale topographic maps and aerial photos) and field trips, has been used as the reference base. Methodology of satellite monitoring of landscapes has been continuously improved by integrating geographic information systems (GIS) into the classification procedure.

The main goals in carrying out satellite monitoring of Estonian landscapes are: i) quantifying the changes in land use and landscape diversity, ii) bringing forward ongoing trends in landscape and give future state prognosis, iii) integrating satellite remote sensing and GIS in spectral-based land cover/land use mapping.

Applied landscape satellite monitoring activity is based on relevant hierarchical scheme (Aaviksoo & Muru, 2001) which consists of 8 landscape types, 21 land cover classes and 58 land cover types (habitat) and has been developed on the basis of Estonian natural and socio-economical peculiarities.

3558 km² or 8% of Estonian territory has been monitored. After 7 years of monitoring work, one can bring forth the following main trends in monitoring areas (and in Estonian nature as a whole):

- i) widespread afforestation,
- ii) increasing of coniferous stands in forests,
- iii) decreasing of clearcut in core areas and increasing in buffer zones of monitored sites,
- iv) increasing of grassland at the expense of arable land,
- v) increasing of fallow land at the expense of abandoned fields and cultivated grasslands,
- vi) overgrowing of natural grasslands and fallow land with shrubs and young trees.

Natura 2000 monitoring

– can we satisfy national needs and EU demands at the same time?

Johan Abenius, Naturvårdsverket

Abstract

Swedish conservation strategies

- Shifting perspectives – from “save the last remnant” to landscape ecology.
- The Swedish model for forest protection – a combination of strict protection of high quality areas and modified use of the “everyday” matrix landscape.
- Does it work? Seems reasonable – but is really an experiment without a control. It is still easier to get political support and resources for acquisition of land rather than for monitoring the effects of conservation measures on biota. The forecast however is that the focus on national environmental objectives will eventually lead to a change in this attitude.
- The national monitoring programme is steering towards biodiversity-relevant parameters.

Conclusion (1): There is a strong national need to deliver information on the state of biodiversity in general and the effects of conservation measures in particular.

Habitat directive demands

Six year reporting cycle. The conservation status as well as the most important results from monitoring activities should be reported.

Conclusion (2): We have to prove that the desired conservation status is reached for each of our habitats and species in each of the three biogeographic regions.

Natura 2000 in Sweden (so far)

From initial phase “Get it done and get back to work” to integration into national conservation.

Conclusion (3): It is here to stay. Natura 2000 is a substantial part of the national protected network.

Condition monitoring

What kind of monitoring? It has to be relevant for conservation objectives. We looked into the condition monitoring system developed and tested by CCW in the LIFE project “Habitat monitoring for Conservation management and reporting”.

Conclusion (4): The CCW concept is an attractive model for monitoring. It is designed to feed data into an aggregated reporting system such as the Natura 2000. We should try to modify it to suit our particular natural environment.

Upcoming question

Will the Natura process be able to accommodate a changing knowledge base? Swedish position: the international commitments should be flexible to future changes.