NINA Report 195

Habitat modelling - A tool for managing landscapes?

Report from a workshop held in Sunnersta, Sweden, 14-17 February 2006

Scott M. Brainerd
Leif Kastdalen
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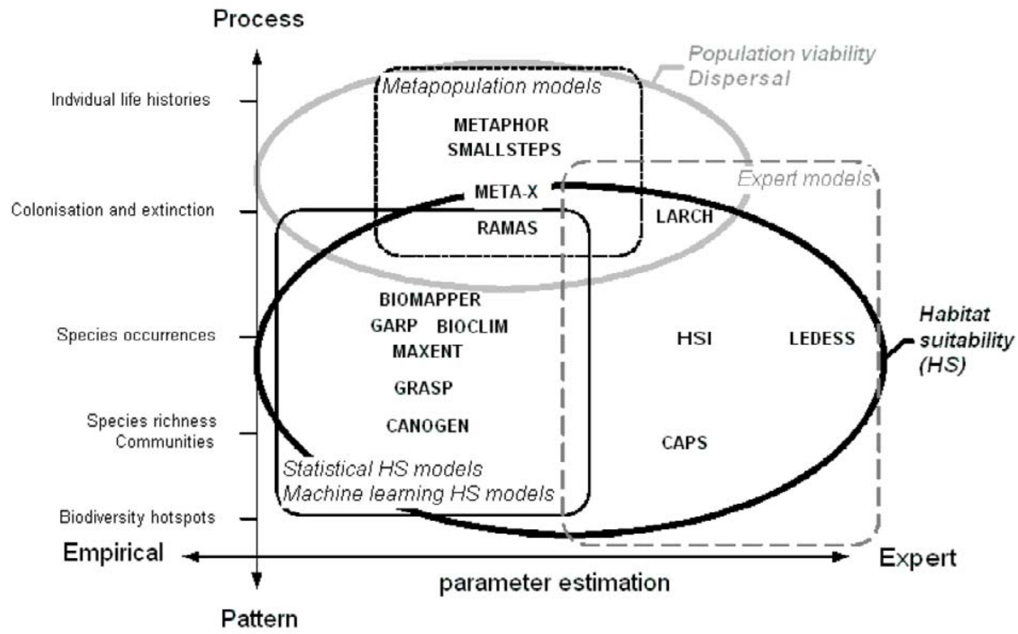
(Editors)



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Oslo, June 2007

ISSN: 1504-3312
ISBN 978-82-426-1755-2

AVAILABILITY
Open

PUBLICATION TYPE
Digital document (pdf)

EDITOR
Scott Brainerd

QUALITY CONTROLLED BY
Erik Framstad

SIGNATURE OF RESPONSIBLE PERSON
Research director Erik Framstad (sign.)

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COVER PICTURE
Figure: Mikael Gontier (p. 10, this volume)

KEY WORDS
Habitat modelling, Geographic Information Systems, GIS, remote sensing, biodiversity, conservation, landscape, nature planning, resource management.

NØKKELORD
Habitatmodellering, GIS, fjernanalyse, biodiversitet, landskap, arealplanlegging, naturforvaltning.

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Abstract

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A sustainable landscape management requires tools for assessment of impacts caused by changes in land use, infrastructure and human settlement. These tools must help to compare alternative scenarios and evaluate their consequences for e.g. biodiversity. Habitat suitability models, based on empirical or expert knowledge can provide such tools. Various types of models have been developed and are used in ecological sciences already. However, their implementation in regional planning in Scandinavia is still very limited.

The Mistra program INCLUDE, together with the Norwegian SatNat program, a user program financed by the Norwegian Space Agency and the Norwegian Directorate for Nature Management (DN) organized a workshop on applied habitat modelling which was held during February 14-17, 2006 at Sunnersta Herrgård in Uppsala. A total of 23 experts from Norway and Sweden attended this workshop and discussed different modelling approaches and how these can be improved and implemented.

It was concluded that the dialogue between researchers and users needs considerable improvement. Researchers must learn more about user requirements, whereas users need a better understanding of the possibilities and constraints in modelling tools. Model validation and quality control are necessary requirements to be met prior to implementation. Relevant background data must be made easily available. Developments in remote sensing techniques and satellite imagery have already produced highly improved landscape information, although there is still a need for improved access to biological data. In addition, better knowledge is required regarding parameters which should be included in modelling approaches. Web-based metadata on spatial and biological data could help to make these more widely available. However, applicable tools for landscape management must combine biological data with data from the disciplines of economics and social sciences. The participants of this workshop propose therefore a series of interdisciplinary seminars on landscape modelling to be organized and held by interested end-users.

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Sammendrag

Brainerd, S.M., Kastdalen, L. & Seiler, A. (eds.) 2007. Habitat modelling - A tool for managing landscapes? Report from a workshop held in Sunnersta, Sweden, 14-17 February 2006. – NINA Report 195. 81 pp.

En bærekraftig forvaltning av landskapet forutsetter metoder som hjelper med vurdering av forandringer i biotoper, bruk av utmark, infrastruktur og bebyggelse. Forskjellige utviklingsscenarier må kunne sammenlignes med hensyn til deres betydning for bl. a. biologisk mangfold. Habitatmodeller, bygd på empiriske data og/eller på ekspertkunnskap, gir oss disse mulighetene. Ulike typer av habitatmodeller er tilgjengelige, og modellering har allerede fått en bred anvendelse i økologisk forskning. Derimot er slike modeller lite brukt i arbeidet med fysisk planlegging og regional forvaltning.

Det norske SatNat-programmet og det svenske Mistra-programmet INCLUDE organiserte et arbeidsmøte som omhandlet anvendt landskapsanalyse og habitatmodellering den 14.–17. februar 2006 på Sunnersta Herrgård i Uppsala, Sverige. På dette møte diskuterte 23 eksperter fra Norge og Sverige ulike spørsmål vedrørende praktisk bruk og tekniske tilnærminger i analyser av habitatmodeller.

En viktig konklusjon fra møtet er at dialogen mellom forskere og brukere må forsterkes vesentlig. Forskere trenger bedre forståelse for brukernes behov, mens brukerne trenger økt kunnskap om muligheter og begrensninger i modelleringsverktøy. Kvalitetssikring og validering av modeller bør være åpenbare krav. Tilgjengeligheten av relevante bakgrunnsdata må forbedres ytterligere. Mens økt tilgang til digitale fly- og satellittbilder og forbedrede analysemetoder har ledet til betydelig bedre landskapsinformasjon, er de biologiske dataene og parametrene som skal inkluderes i modellene, ofte ufullstendige eller ukjente. Internettbaserte meta-databaser kan bidra til å gi et overblikk over eksisterende data. I utviklingen av anvendbare verktøy for landskapsforvaltning kan de biologiske modellene også kombineres med økonomiske og human-økologiske modeller. Deltagerne på arbeidsmøtet anbefaler derfor en tverrvitenskaplig seminarserie om landskapsmodellering.

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Contents

Foreword

The Swedish program INCLUDE and the Norwegian program SatNat held a 3-day workshop on applied landscape analysis and habitat modelling in Uppsala, Sweden in February 2006. This meeting was financed by the Swedish Mistra research program, the Norwegian Space Agency (NRS), and the Directorate of Nature Management (DN). The Norwegian Institute for Nature Research (NINA) also contributed funds and was responsible for coordinating the meeting and producing this report.

At the workshop, invited experts from Swedish and Norwegian management agencies and research institutions discussed selected questions regarding habitat modelling techniques and practical application and relevance of these. Although the thematic focus centred on biological diversity and species ecology, the overall goal was to expand modelling efforts to include economic, social and cultural aspects. The workshop also aimed to promote the use of remote sensing data and modelling in physical planning and impact evaluation relative to infrastructural projects and/or spatial planning. Models that can simulate and evaluate landscape changes through the use of remote sensing and expert knowledge are important tools in sustainable development.

The workshop addressed several important issues on the topic of habitat modelling and its applications in the real world. Emphasis was placed upon how and to what extent habitat and landscape models can be applied to the needs of managers and planners. The discussion on types and quality of available data focused on the need to balance detail and resolution with robustness and generalization for management applications. The advantages and drawbacks of expert and empirical models were contrasted, and optimal analytical methods were discussed in this context. The broader issue regarding the hitherto limited application of models was discussed and analysed, with a view to improving communication and interaction between modellers and managers to ensure more effective and informed decision processes. In addition, participants discussed the need for incorporating human-related (social, cultural) values and valuations as parameters in the same fashion as ecological parameters in applied models for management and planning.

This meeting is one of a series of workshops in the INCLUDE and SatNat programs, which aim to improve dialogue between managers and researchers in order to derive optimal application of knowledge in the sustainable management of biodiversity and human activities. This report provides an overview of the topics presented at the workshop, and the conclusions of the work group discussions. We hope that results of this workshop will contribute to a better understanding and application of models in practical management in planning in Scandinavia, to the benefit of the environment, and ultimately, ourselves.

Oslo, Grimsö, June 2007
Scott M. Brainerd, Leif Kastdalen, and Andreas Seiler

1 Introduction

Spatial planners are obligated to anticipate, and where possible, mitigate the effects of development projects on the environment. This requires sufficient knowledge regarding the habitat requirements of species of flora and fauna that can be adversely impacted by such measures. By incorporating our knowledge of the habitat needs of species into spatial planning, we can identify important habitat and landscape features for species or communities of species of special concern. Once these features have been identified, managers can use these data in the decision-making process in order to avoid or mitigate potentially negative ecological impacts (see Chapters 3, 4, 12, and 15 in this report). Such data can also be used in general conservation planning to identify predicted expected distributions of species of concern. This includes potential “hot spots” or areas with unique attributes that are attractive for clusters of red-listed species (see Chapter 5). Information can then be used for focusing mapping and monitoring efforts in such places. At a broader scale, habitat modelling can be a cost-effective tool when combined with remote-sensing data and computer-based mapping technology for identifying important landscape and habitat features. It can also be used to predict species distributions and the consequences of different human activities upon them, and this is probably its most important applied context (see Chapter 6).

In this report, Gontier (Chapter 2) provides an excellent overview of spatial ecological models and how these can be used. There are two basic types of habitat models: empirical and expert models. Empirical models build upon data derived from field investigations of the species being modelled. Data complexity can vary from simple (presence/absence) to complex (habitat preferences and/or landscape-scale effects). Such models are limited by the amount and quality of available data, and the applicability of such data to local situations. Expert models are generally simple, and are based on subjective evaluations by biologists with extensive knowledge of species habitat requirements (see Chapter 10 and 11). In cases where empirical data are lacking, expert models can serve as a useful tool where general knowledge on species and their habitat preferences are available. However, expert models are limited by gaps in knowledge and are often only applied locally.

Geographic Information Systems (GIS) are ideal platforms for integrating our knowledge of species distributions with map data in order to develop tools spatial planners need for making responsible decisions regarding infrastructural projects. In addition, GIS can be used as an important tool in the systematic mapping of ecosystems and the identification of gaps between present and desired amounts of land coverage types needed to meet specific biodiversity goals (see Chapter 13). Such tools can contain indicators, thresholds and standards which can help identify important habitat features for species of special concern. In Sweden and Norway, remote sensing data and vegetation map coverage and classification systems are highly developed. There is, therefore, a great potential for using these data in the development of increasingly detailed map models for a wide range of species for which empirical data exist or can be gathered (see Chapters 7 and 8). Indeed, combined map data can be enhanced using innovative techniques (see Chapter 9). Advanced modelling techniques and methodologies can be applied to enhance our understanding of species spatial requirements in the landscape on the basis of empirical data regarding habitat requirements and/or movement patterns (see Chapters 12, 14, 15, 16, 17, 18).

In this report, the authors present an overview of current status regarding the application of habitat modelling as a tool in conservation and planning in Scandinavia. We identify important tools and concepts, as well as needs for the future. There is a great need for systematizing and standardizing data and methodologies, and consolidating efforts between our countries and on a global scale to ensure the most efficient use of existing knowledge and resources. This report and seminar represent a first step here in Scandinavia. We hope that the information and ideas presented here will serve as a foundation for future work and cooperation in this important field in the years to come.

2 Spatial ecological models - an overview

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2.1 Introduction

Spatial ecological models have been developed within different research disciplines like landscape ecology, spatial ecology and conservation biology (Guisan & Zimmermann 2000, Akcakaya 2001, Opdam et al. 2001, Scott et al. 2002). Many of these are or could be implemented in a GIS interface allowing numerous applications. These models differ from each other in many ways but also share a lot considering the technologies and methods that are used. It may therefore be interesting to propose some classification of the models in order to see in which ways they differ or resemble each other. To understand the characteristics of the models, as well as their similarities and differences, can aid in model selection for a specific purposes.

2.2 Classification and characterization of spatial ecological models

There are different ways to classify ecological models. An attempt to classify such models is presented in **Figure 2.1**.

It is important to remember that there are no sharp boundaries between one type of model and another and the classification that is presented in **Figure 2.1** shows gradients between different extremes. One way to classify them is to take into consideration the modelling techniques and statistical methods that are used. In an attempt to plot some of these models, this information can form one axis, where the two extremes are defined by more process-based models on

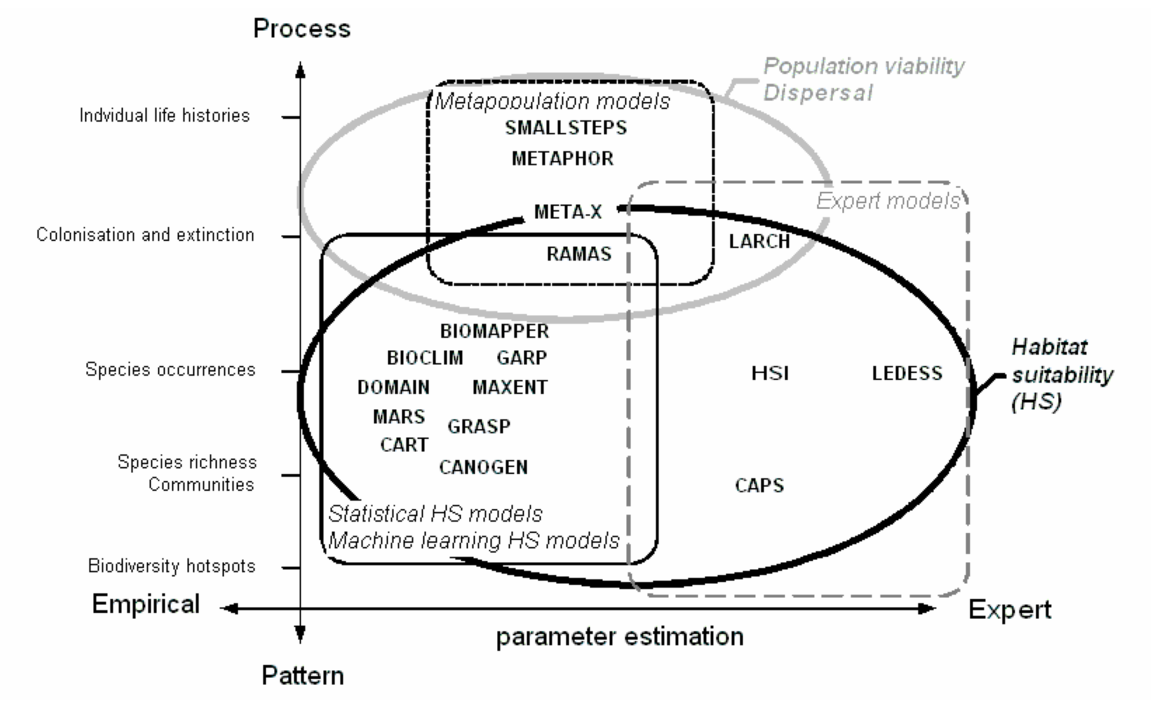

Figure 2.1 *Spatial ecological models (Adapted from Gontier et al. 2006)*

the one hand and more pattern-based models on the other (**Figure 2.1**). Process-based or mechanistic models require detailed information on underlying causal mechanisms of the response variable (Gontier et al. 2006). Pattern-based or phenomenological models intend to uncover relationships between variables and observed phenomena without trying to explain the mechanisms involved in the response. Another distinction between the models that forms a second axis in **Figure 2.1** is the distinction expert models and models requiring empirical data. Expert models such as LEDESS (Knol et al. 1999) require a high level of knowledge on the subject that is modelled, whereas on the other end of the axis some empirical models such as GARP (Stockwell & Peters, 1999) or Biomapper (Hirzel et al. 2002) can be run without specific knowledge requirement on the subject. In the space defined by these two axes it is possible to distinguish a number of categories and families of models. One distinction is between habitat suitability models and those pertaining to population viability and dispersal. Habitat suitability models provide distribution maps of occurrence probabilities and population viability and dispersal models calculate population dynamics and viability of populations (Gontier et al. 2006).

Other criteria could be used to describe or classify spatial ecological models. Within empirical models, the type of input data that is required can help to differentiate models. Some only require species presence data, whereas others require both presence and absence data. Finally, other methods may require abundance data. Another way to characterize empirical models is through the type of algorithm on which the calculations are based, whether these are statistical (e.g. logistic regression, canonical correspondence) or machine learning algorithms (e.g. artificial neural network, maximum entropy). The data requirements (type of data, format, need for specific variables) for the independent variables can also vary between models. Some models do not accept categorical data (e.g. Biomapper) whereas others do (e.g. Maxent, Phillips et al. 2004). Further, the geographical scale at which the model is implemented can have consequences for data needs and availability. The purpose for which a specific model was developed can also be relevant. The user friendliness varies between models from “research oriented” models where advanced knowledge on the method is required to ready to use software packages. The lack of specific software may in turn allow more flexibility in the implementation of the model. Finally, the type of results provided by the model could be a determining factor. Some models provide stochastic results (e.g. GARP) whereas others are deterministic. Moreover, the output of the modelling also varies, with some models providing binary maps and other continuous ones thus inducing differences in the interpretation.

2.3 Conclusion

The variety of spatial ecological models is constantly growing and it may be difficult to understand the differences and similarities between these. To provide a classification of these models is a difficult exercise since they are belonging to different research fields and there is no common terminology to characterize them.

The choice of a spatial ecological model may be driven by many factors. The aim of the study may be the first priority, but other realities such as data requirements, the type of results or the user-friendliness of the model may also determine selection. It is important to remember that the diversity of existing models offer a wide range of applications useful for solving a large variety of problems.

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3 Tools for reliable and transparent predictions in environmental assessment

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3.1 Introduction

Endeavours to achieve a sustainable society require instruments to investigate the impacts of development actions. Environmental impact assessment (EIA) and strategic environmental assessment (SEA) are universally recognised planning instruments which are used in decision-making and for formulating development actions, and in sustainable development (Glasson et al. 2005). According to the EU Directive on Environmental Impact Assessment (Official Journal of the European Communities 1997), EIA applies to a large number of projects including the development of infrastructures for transport, water and other supply systems. However, initial decisions on urban expansion and major infrastructure investments are often made at the strategic stage and for these situations EIA regulations cannot be applied. Instead a strategic environmental assessment (SEA) can be prepared, which addresses the environmental impacts of strategic decisions (Balfors & Schmidtbauer 2002). The EU Directive concerning the assessment of effects of certain plans and programs on the environment (Official Journal of the European Communities 2001) meets the need for environmental assessment of strategic decisions and is directed towards such sectors as transportation, agriculture, forestry and town and county planning.

According to Sadler & Verheem (1996) SEA can be defined as: *´a systematic process for evaluating the environmental consequences of proposed policy, plan or programme initiatives in order to ensure they are fully included and appropriately addressed at the earliest appropriate stage of decision making on par with economic and social considerations´*. The main rationale for applying SEA is to help to create a better environment through informed and sustainable decision-making (Fisher 2003). In addition, SEA contributes to a more effective promotion of sustainable development and allows for the assessment of cumulative impacts (Sadler & Verheem 1996, Therivel & Partidario 1996, Glasson et al. 1999). However, the high level of abstraction of policies, plans and programs involves major methodological problems for the prediction of impacts (Hilden et al. 1998).

In this contribution to the INCLUDE workshop, I discuss the need for relevant data and adequate prediction tools in EIA and SEA. This discussion serves as an introduction to the later contributions on landscape ecological modelling in SEA.

3.2 Environmental goals

In Sweden, the basis for environmental and nature protection policies consists of 16 environmental quality objectives. The environmental quality objectives describe the quality and the state of the environment and natural and cultural resources of Sweden, which the Parliament judges to be environmentally sustainable in the long term (The Swedish Environmental Objectives Council 2006). The environmental goals provide a national framework for planning and decision-making, and serve as guidance for local and regional actions. Local and regional authorities need to define objectives which serve to implement national objectives.

In order to achieve national objectives and their local and regional equivalents, consistent application in planning and decision-making is required. Within EIA and SEA, the environmental objectives are a central part of the terms of reference for the assessment of different development alternatives. The predictions should thus provide output that can easily be related to the environmental objectives.

3.3 Impact prediction in Environmental Assessment

Scoping and the prediction of impacts are two core steps in EIA and SEA. Scoping consists mainly of three parts – to identify key issues and impacts, to consider alternatives and to find forms for the involvement of the public and other stakeholders. This involves that the agenda for the EIA or SEA is set within the scoping process. The objective of impact prediction is to identify the magnitude, significance and other dimensions of identified change in the environment resulting from a project or action (Glasson et al. 2005). Hence, the outcome of the impact prediction should provide relevant data in a sense that it focuses on the most significant issues and supports the comparison between different development alternatives.

Until recently, EIA has mainly focused on relatively small-scale and local effects (Treweek et al. 1998, Geneletti 2002) and has often concentrated on protected areas and species (Byron et al. 2000). Also Gontier et al. (2006) conclude that today´s EIA practice shows a lack of consistent quality in current biodiversity assessments. Most Environmental Impact Statements consider species and local habitats even though they are often restricted to protected species and protected areas. Furthermore, ecosystems are rarely considered (op. cit.). Some limitations linked to the nature of the EIA process are the narrow time frame and the imposed physical boundaries of the projects (Treweek 1996). However, the implementation of the SEA process offers opportunities to take into account impacts across administrative borders, cumulative effects, widespread off-site impacts (Treweek et al. 1998) and to consider scales of ecological processes (Balfors et al. 2005).

The selection of impact prediction tools is thus an issue of major importance for the quality of the EIA and SEA. The methodological approach should be scientifically correct so that the output of the predictions is accepted by experts and public. This requires research efforts in order to identify an appropriate approach. In addition, the prediction tools should deliver differentiated forecasts over time and space, which facilitate the analysis of environmental changes at different stages and places after the implementation of the proposed development. In this context, Geographical Information Systems (GIS) provide an effective tool as they allow quantitative assessments that take into account both spatial and temporal scales (Gontier et al. 2006).

Besides the methodological aspects, other issues need to be considered regarding the selection and application of prediction tools. In order to be effective tools in planning and decision-making there is a need for willingness to apply the tools in planning practise. Furthermore, relevant competence for the application of the tools needs to exist or to be developed. Without adequate knowledge, no prediction tool will be effective. Finally, there should be an incentive to apply tools. Without incentives it will be difficult to encourage professionals to apply ecological models.

3.4 Data and scale in environmental assessment

In order to conduct an effective and useful environmental assessment appropriate data are needed that add relevant information on the environmental impact of the proposed activity. This rises the question of what type of data are needed in EIA and SEA, and how much data is enough? Basically, the information should facilitate an adequate analysis of significant impacts and highlight the main differences between different development options. Furthermore, due to limited time and resources to carry out an EIA or SEA, it is important to focus on the main issues and to apply the tools that generate the data that are needed.

The selection of data and tools relates to the scope and the scale of the assessment. Each tier of the planning process deals with certain types of issues which are discussed at different levels of detail. Thus, different data are needed in EIA and SEA. Therefore, it will be relevant in the further development of SEA to identify the types of data needed for different sectors and levels. This implies that the assessment of biodiversity issues for local activities focus on other issues that require other tools and data than the analysis of biodiversity impacts at the regional or national level. Yet, the interconnection between these levels should not be ignored as local activities have bearing on the regional and national environment and vice versa. Hence, the scope and the scale of the SEA do not need to be restricted to a particular level, but instead multiple scales or range of scales appropriate for the analysis can be considered (Azcarate & Balfors 2006).

3.5 Approaches for data collection and objective formulation

Two types of SEA systems are widely cited in the SEA literature; these are baseline-led SEAs and objective-led SEAs. Therivel (2004) defines a baseline-led SEA as a distinct environmental yardstick of discrete SEA themes, objectives, and indicators that are used to describe the baseline environment, identify problems, and influence the objectives of a strategic action. On the other hand, in objective-led SEAs, sustainable objectives for strategic actions are developed and the different alternatives are tested through the use of indicators to see if these objectives can be achieved (op. cit). Objective-led SEAs are also thought to help reduce the need for baseline data by focusing the SEA process on preset issues. However, objective-led SEAs could have the effect of excluding important issues from environmental assessments and may even lead to biased appraisals (Azcarate & Balfors 2006). By using both types of SEA systems jointly, first by setting preliminary objectives and then by revising them as sustainability issues are identified, an iterative, effective, and context relevant data collection process can be established that could aid in solving the current data collection challenges accompanying SEA (Azcarate & Balfors 2006).

3.6 Conclusions

Impact prediction is a crucial part in EIA and SEA that is of major importance for the quality and effectiveness of assessments. In order to make adequate predictions on biodiversity impacts, tools and data are needed that generate relevant information which account for relationships between species and habitats, local and regional scales, and time and space. In addition, tools should meet the overall EIA and SEA requirements regarding transparency and reliability, and thus be adapted to those who use them: experts, decision-makers and the public. This creates a challenge for researchers who develop the tools and practitioners who conduct the assessments.

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4 Habitat modelling and nature management

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4.1 The meaning of nature management

We see the management of nature or the landscape as a management of the natural resources. I like to view the management in general as a triangle, with the national parks and nature reserves on top. Below them I put the protection of shorelines, and under these we have possibilities with the help of SEA, EIA, spatial plans - near the base line "special considerations" and on the base line "general considerations", stipulated in the law, e g forestry, road building and plan- and building acts. Some laws and conventions cross the whole triangle, like the environmental code and the European Landscape Convention. On the top of the triangle it is more an active care where environmental agencies pay and in the bottom other sectors pay and take consideration into nature.

4.2 Three important points when using models

1) Reasons and possibilities for good models

- <u>The public</u> can and must be involved – and then we can get the support among the public that we work hard for during the last years,
- Alternatives are important – and with models made by help of technical facilities it is possible to get <u>alternatives presented in equal ways,</u>
- You can adjust <u>modelling</u> – it means that if you change income data you can see how sensitive they are (sensitivity analyses)

Data influences the results … therefore it is important to use the right data!

2) The most basic and important question: What is the problem?

Try to find the solution as early as possible in the chain. (An example, we must build a new road - Why? Because a lot of people have to go to school and work in this city … But maybe we can establish schools and jobs in the city where people live… I mean the real problem is not that the road is too narrow, but rather that the jobs are in one place and people live in another)

3) Try to take a holistic approach

A visionary model should be one with a lot of different inputs where I, as a manager, can see different results depending on how I moderate my inputs in a holistic approach. As a manager you should see your work in a holistic point of view or/and how to put your model in more complex systems.

Sustainable development = ecology + social + economy

4.3 Nature Management at EPA

If we look at the work at Environmental Protection Agency (EPA) with a focus on my section (the Section of outdoor recreation and physical planning) I will shortly present some projects/duties under following five headlines

Valuation, inventory More general or thematic inventories, some times with following programs or action plans. For example, with regard to county nature programs. In this county of Uppsala we are just now examining this within the context of an outdoor recreation project.

Special cases of exploitation etc Assessing the impact on the nature and the landscape, check/assess the documents and the environment impact assessment (EIA), e g roads, railways and wind power stations.

Comprehensive planning Strategic Environmental Assessment (SEA), landscape analysis, the SEA handbook and a report on different landscape methods of landscape analysis.

Implementing of conventions The European Landscape Convention (ELC), together with the National Heritage Board dealing with international work and with implementing ELC into Swedish legislation, and as a result of the convention a pilot project at seven County Administrative Boards to carry out regional landscape strategies. The EPA has a role of supporting and in the end writing guidelines of landscape strategies.

Methods, projects “Good examples” and guidelines, in cooperation with Nordic Council of Ministries, the National Heritage Board, the National Board of Housing, Building and Planning and others, such as the Nordic Landscape, which is a basic document for ELC in the Nordic countries, “Tvärs” (“Inter”) integration nature-culture on regional and national level. And thinking in the light of ELC, how to use landscape as an arena for communication experts-public-politicians (INCLUDE ?).

4.4 To integrate a lot of interests, laws, guidelines and programs

A complex world of planning and decisions: The Environmental Code, areas of national interests (for example for nature, recreation, tourism, transport, energy), regional development program (RUP), regional transport infrastructure plan (LTP), integrated coastal zone management (ICZM), etc

In the county of Stockholm there are ongoing discussions and different plans concerning a new road link to Stockholm. Good tools could be an <u>integrated spatial plans with SEA</u> – the regional plan – and models with the public involved as well as different objectives – satisfaction of the objectives.

5 The need for knowledge in conservation of "threatened" species

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5.1 Introduction

In order to ensure that "threatened species" are included in the conservation of our biological diversity, it is necessary to 1) prioritize species, and 2) manage landscapes or implement measures for prioritized species. Point 1 includes the development of Red Lists with consequent prioritization of species (e.g. "responsibility species"), while Point 2 relates to the daily landscape management which often is conduced at quite local levels (landowner, municipality) as well as special measures (management plans) which often are initiated on the basis of national policy.

Conservation of threatened species requires efforts and knowledge within several themes, and each theme will require varying levels of information. I present comments and evaluations of the need for knowledge for these two aspects: 1) prioritization of species, and 2) management of prioritized species.

5.2 Species prioritization

A Red List is a critical basis for the formulation of management priorities. A regional Red List that has been formulated according to World Conservation Union (IUCN) criteria is an evaluation of the probability that species will become extinct for a specific region (in this case, Norway) - nothing more and nothing less. Categories and criteria for red-listing species are given by the IUCN (2001), with regional (in this case national) guidelines for classification given by IUCN (2003), with updated guidelines available on the IUCN web site (latest version updated April 2005).

IUCN has specified a set of criteria for evaluating extinction risks (establishment of Red Lists) for species under consideration. These include knowledge on distribution and abundance, ongoing changes in these two variables, knowledge on population structure (geographic distribution and degree of fragmentation) as well as information on species ecology (generation time, dispersal ability). It is important to understand that a species can appear on the Red List even though it is very abundant if signs of decline are strong enough (IUCN A-criteria). Thus, knowledge regarding population change for our common species is very relevant for the Red Listing process.

Much of the kind of knowledge needed here is quite lacking for most of the roughly 20,000 species for which we are now conducting detailed evaluations for the new Norwegian Red List which shall be presented at the end of 2006. The reason that such an evaluation can still be conducted is due to the fact that the IUCN has established a set of knowledge levels upon which Red List evaluation can be based. This includes the entire spectrum which run the gamut from direct estimates (e.g. through population viability analyses) to indirect knowledge and assumptions at the other extreme. IUCN specifies that it is important that classifications that employ less precise knowledge be well documented. In other words, indirect knowledge and the assumptions used must be described and be available for everyone with an interest in the background for evaluations.

Proper documentation is essential since indirect knowledge is being used in evaluating most species on the new Red List, and will also be used far into the future. A typical usage of indirect knowledge combines information on a given species' habitat along with knowledge regarding the availability and changes in this habitat. In this regard statistically reliable and representative knowledge on landscape parameters is important. A primary challenge for us in Norway is that we only have such information for commercial forest (Norwegian Forest and Landscape Institute), and these data lack the detail and variables needed for evaluating many species.

In uncertain cases (e.g. when indirect knowledge and assumptions are used), the IUCN states that a precautionary but realistic approach should be undertaken, i.e. that a somewhat lower risk-tolerance should be employed, whilst avoiding a "worst-case" scenario. Given such an approach, if the species is still classified as a Red List species, it should be placed in the highest appropriate category. Such an approach will enable more species to be included on the Red List than would otherwise be the case if we had adequate data.

5.3 From Red List to management priority

Evaluation of extinction risk and prioritization of management measures are two related but different processes (p. 5, IUCN 2003, p. 11 IUCN 2005). A Red List provides an estimate for extinction risk for a taxon. This risk can be an effect of human activity, but it can also be due to natural processes. For example, a species may be at the limit of its northerly range limit in southernmost Norway but be otherwise abundant further south in Europe. Prioritization of management measures should therefore take into account other aspects. Such an aspect may include the proportion a Norwegian species population comprises of the total global or regional population (the status of a given taxon in a global perspective), the potential range of the species in Norway, and other things including costs, logistics, possibilities for success, and other biological characteristics for the species. It should also be mentioned here that the IUCN states that regional authorities can decide which taxa can be excluded from regional Red Lists if its proportion of the global population is quite low (< 1 %, see p. 10, IUCN 2003) for a given region. However, it is up to each country to determine how its Red List will be used. Sweden has stated that biological diversity in its entirety is to be conserved, and thus all species on its Red List are prioritized.

5.4 Species management and spatial planning

For most species that will be included in the Norwegian Red List 2006, it will be impossible to directly map the ranges of all those which should be given priority for spatial management and planning due to limitations in both technology and knowledge. For these species, it is therefore natural to focus efforts on the conservation of important habitat and landscape features associated with these species. This requires, however, that we have detailed information regarding given species' habitat use and to what degree these tolerate habitat change, which is not always the case.

Knowledge regarding the location of "hot spots" (i.e. areas which contain many Red List species) is important for spatial management and the conservation of these. This approach is utilized in the program Environmental Registration of Forest (MIS) and is also in the process of being established for the management of commercial forest in Norway.

For many species, it will hopefully be possible to develop habitat models that can predict expected distributions of species. This will be a useful tool for both spatial planning and for Red Listing evaluations. Such models can be developed by combining observations of species with diverse kinds of spatial information. For terrestrial ecosystems, such spatial information can include spectral sets from satellite images, topography, geology, soil types, climate (temperature, precipitation), etc. Such models will then calculate probabilities for finding particular species or species communities at fine geographical scales. The precision of these models must then be tested through field investigations before they are used for landscape management purposes.

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6 GIS-based habitat models in spatial planning

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6.1 Introduction

In order to achieve sustainable development, biodiversity issues need to be considered in spatial planning. Spatial planning refers to the methods used by the public sector to influence the distribution of people and activities in spaces of various scales. Spatial planning includes all levels of land use planning including urban planning, regional planning, national spatial plans, and, in the European Union, international levels (CEMAT 1983). It is at the same time a scientific discipline, an administrative technique and a policy developed as an interdisciplinary and comprehensive approach directed towards a balanced regional development and the physical organisation of space according to an overall strategy.

Spatial planning concerns the use of land, water and the built environment from ecological, social and economical aspects. It involves for instance the planning of infrastructure and the comprehensive and detailed development plans of the municipalities. Furthermore, spatial planning involves Environmental Impact Assessment (EIA) for projects (Official Journal of the European Communities (OJEU) 1997) and Strategic Environmental Assessment (SEA) concerning certain plans and programs (OJEU 2001).

As a consequence, spatial planning affects biodiversity, and in order to integrate biodiversity issues in spatial planning, the spatial dimension needs to be considered. In Sweden, the environmental quality objectives constitute a foundation for the integration of, among other things, biodiversity issues in spatial planning (Government prop. 1997/98:145, 2004/05:150, Naturvårdsverket & Boverket 2000). In addition, the environmental objectives provide the basis for the formulation of local environmental objectives and interim targets and/or indicators.

For the integration of biodiversity issues in spatial planning, a methodological framework for Landscape Ecological Analysis and Assessment (LEA) has been developed (Mörtberg 2004, Mörtberg et al. 2007). The methodology involves the identification of relevant environmental objectives, of the main threatening processes and environmental problems, followed by the identification and/or formulation of biodiversity targets relevant for the particular planning situation, and of suitable indicators related to these targets. One way of deriving indicators or detailed interim targets is to use ecological profiles (Vos et al. 2001). They consist of functional groups of species, with habitat requirements defined along a gradient of specialisation, resource requirements, dispersal capacity and other critical properties.

When suitable indicators have been identified, spatial predictions of suitable and accessible habitat for each selected ecological profile can be performed using GIS-based habitat models (Mörtberg et al. 2007, Gontier et al. 2006). The GIS environment has the advantage of being suitable for scenario testing, which means that planning scenarios such as alternative exploitation plans and projects, and management options can be tested on ecological profiles, on a landscape level. The last step in the LEA methodology is the assessment, where effects of the scenarios on the indicators are interpreted and assessed compared to the targets. From here, iteration of the scenarios and mitigation measures can be performed on the scenarios, and tested again. Finally, the scenarios can be evaluated and the LEA process and results can be used as decision support (Balfors et al. 2005, Mörtberg et al. 2007).

6.2 Applications of GIS-based habitat models in spatial planning

Urbanisation scenarios for Greater Stockholm
With the LEA as a framework methodology, several studies and practical applications in spatial planning have been conducted using GIS-based habitat modelling as a key tool. One example is the study of different urbanisation scenarios on a regional level in the Greater Stockholm area. Within this study, three scenarios for future urbanisation were tested, with different principles for localisation of 250,000 new households in the year 2030, which were developed by the Office of Regional Planning and Urban Transportation (ORPUT 1995). They consisted of the Scenario Dense, with a dense urbanisation pattern within walking distance from the subway, the Scenario Diffuse, with urban sprawl scattered over the whole area, and Scenario Infra, with urban settlements within walking distance of long-distance commuter trains.

The ORPUT (1995) evaluated the effects of the scenarios on transportation, while Mörtberg et al. (2007) evaluated the effects of the same scenarios on two biodiversity targets for forest in the region. These targets were A) intact large forest tracts with characteristic combinations of coniferous forest and wetlands, maintaining habitat for two forest grouse species, both resource-demanding and one dispersal-limited, and B) intact networks of coniferous forest in the inner suburbs, maintaining a community of non-urban, sedentary forest tits. GIS-based habitat models were used for the purpose, in the form of regression models, similar to the empirical GRASP model in Gontier et al. (2006). Further, spatial autocorrelation was taken into account (Mörtberg & Karlström 2005). The GIS-based habitat models incorporated variables representing habitat quality, quantity and location, and habitat quality included vegetation parameters as well as urban disturbances such as traffic noise and recreation. The outcomes of the models were spatial predictions of suitable habitat in the present situation and with the three scenarios. The results showed expected results as the Scenario Diffuse had serious impacts on Target A, (large forest tracts) and Scenario Dense had serious impacts on Target B (forest network in suburbs). More interesting were the revealed negative impacts of Scenario Infra on both targets. Furthermore, these negative impacts could be localised and easily mitigated and avoided. In practice, this would mean to leave certain commuter train stations unexploited. The study showed the great potential of this type of GIS-based methods.

The National Urban Park of Stockholm
In the next example, the Stockholm County Administrative Board was commissioned by the Swedish government to develop a program for management and development of the Stockholm National Urban Park. The assignment involved the formulation of targets and guidelines for the long-term management and development of the park. The program should involve all land that can be considered as important for the values of the park, not only within the park itself, and further concern both the conservation and development of those values. As a part of this assignment, the EMA research group together with the Stockholm University performed a LEA in order to reveal the potential for biodiversity associated with the broad-leaved deciduous trees in the National Urban Park, providing habitat for many red-listed invertebrates. The aims were to identify important parts of the ecological network for species linked to these habitats, to evaluate the probable consequences a development scenario of current interest, and to provide knowledge for planning, management and development of the National Urban Park (Mörtberg & Ihse 2006).

Since the target was suitable and accessible habitat for specialised and dispersal-limited invertebrates, the ecological profiles that were outlined included properties such as specialisation on large deciduous trees, dead wood and other habitat characteristics, together with different assumed dispersal characteristics. These detailed ecological profiles could be outlined to such detail, since a habitat map of high quality from a biodiversity perspective could be used (Löfvenhaft & Ihse 1998). Spatial predictions of suitable and accessible habitat for the ecological profiles were derived by GIS-based expert models. From the results, the main characteristics of the ecological networks could be outlined, such as potential core areas and dispersal links. These areas were considered to have the highest potential for long-term development of the broad-leaved deciduous tree habitats of the National Urban Park.

Hanveden
The third example is an ongoing project where a land use plan and strategy is being developed for a large nature area, Hanveden, which is a part of the green structure of Stockholm. The aim of the project is to plan the area as a whole, integrating nature conservation, recreation and forestry. The main actors are four municipalities, owning large parts of the land, and Skogssällskapet, as forest managers of these areas, but the Hanveden area includes also other landowners and conflicting interests. The EMA research group is working with nature conservation planning and strategies in the area, using different kinds of GIS-based habitat models as tools (Gontier & Mörtberg, in prep).

6.3 Discussion

The experience from these and other projects shows that GIS-based habitat models can be very useful in spatial planning. They can be used to support decisions in many planning situations, as for instance assessment of impacts on biodiversity on a landscape level, for the spatial planning of municipalities, such as comprehensive plans, detailed development plans, EIA and SEA, for monitoring and for prioritising resources concerning restoration, compensation and mitigation measures. Questions that can be addressed concern e.g. the localisation of potential suitable and accessible habitat for prioritised species, the localisation of important parts of ecological networks, such as core areas and dispersal links, and further where should exploitation be avoided, where will exploitation be least harmful and where can mitigation measures and compensation be most effective.

However, GIS-based habitat models need to be developed further. Firstly, the spatial and temporal predictions need to be developed within the scientific community of conservation biology (see Scott et al. 2001). Secondly, the use of GIS-based habitat models as decision support in spatial planning is a slightly different task. For both purposes, a critical point is the existence and availability of detailed vegetation or habitat maps, for which techniques already exist. In addition, efforts to use GIS-based habitat models for decision support often reveal knowledge gaps concerning the effects on biodiversity of the particular planning situation. When sufficient data sets exist and are available, empirical models (which need input of empirical data on the studied item) can be very useful.

If the intention is to provide spatially explicit decision support to spatial planning, by others than researchers, efforts will though be necessary to build a knowledge database, preferably gathered within expert models available for planning. A good example is the LARCH model, which is in use in spatial planning in the Netherlands. Much research within conservation biology has been organised to provide the LARCH expert model with parameters, such as home range, dispersal capacity and sensitivity to barriers, size of minimal viable population and density of species in different types of habitats (Vos et al. 2001). This type of effort for answering questions, posed by real-world planning problems concerning biodiversity, could be done for instance on a Nordic level, for countries that share biogeographical conditions.

6.4 Conclusions

- Detailed vegetation/habitat maps need to be developed across landscapes and regions. Remote sensing techniques for this already exist but are often only applied in smaller areas, or are not detailed enough for habitat modelling with high precision. Furthermore, such data need to be available for the scientific community.

- Empirical models need input of data on the item studied, and are limited by the amount and quality of data. However, these are often the only option available since the question posed by the planning situation may not be well studied in science (i.e. urbanisation effects on biodiversity). National and international trends of data-sharing make this type of models more feasible, e.g. Artportalen (SEPA), and Global Biodiversity Information Facility.

- Expert models are today limited by knowledge gaps and scattered knowledge. Here we suggest joint research efforts to gather knowledge, on a Nordic level.

- In the long term, the use of GIS-based habitat models as decision support within spatial planning will need the development of tools for planners, hopefully skilled in biodiversity conservation planning but not researchers. This calls for the development of indicators, thresholds and standards.

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7 GIS and satellite data in Sweden

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7.1 Introduction

Several organizations in Sweden produce and provide nationwide geographic information in digital format in various thematic fields. The presentation is non-exhaustive and the purpose is to give examples and indicate where more information can be found.

7.2 An overview

Four national authorities present their products annually in the document "Kartplan" ("Map plan"[1]). Documents that can be downloaded from this website are in Swedish, but additional information on map services can be found on the web sites of the respective authority:

Lantmäteriet *National Land Survey of Sweden* E.g. basic geographic and land information (including real property), aerial photos, satellite images, historical maps	http://www.lantmateriet.se/
Sveriges Geologiska Undersökning, SGU *Geological Survey of Sweden* E.g. bedrock, quaternary deposits, groundwater, geochemistry, geophysics and mineral and bedrock resources	http://www.sgu.se/
Sveriges Meteorologiska och Hydrologiska Institut, SMHI *Swedish Meteorological and Hydrological Institute* E.g. climate, runoff, surface waters (lakes, rivers, coastal waters and sea),	http://www.smhi.se/
Sjöfartsverket *Swedish Maritime Administration* E.g. nautical charts, hydrographical surveys	http://www.sjofartsverket.se/

The web service ***GeoLex***[2] (in Swedish) provides up-to-date information on several of the products available at Lantmäteriet. The web site allows for interactive use, e.g. zooming, search, retrieval of documentation and metadata. Two examples are given in **Figure 7.1**, one illustrating the coverage of IRF aerial images, altitude 4 600 m, and the actuality (year).

The Swedish Land Cover Data, SMD, is a more detailed land cover product than the European Corine Land Cover. The spatial resolution is better (minimum mapping element 1-5 ha compared to 25 ha) and the nomenclature has been adapted and a number of subclasses have been added to the Corine nomenclature, see Appendix. In **Figure 7.2**, a more detailed example of the SMD is given. At the ***GeoLex*** site, detailed metadata information can be found, divided into squares equivalent to a topographic map. The product is available in vector and raster format. Lantmäteriet has an archive with historical maps, also available through the web service. An example is given in **Figure 7.3**.

[1] http://www. lm. se/kartplan/index. htm.
[2] http://www. geolex. lm. se

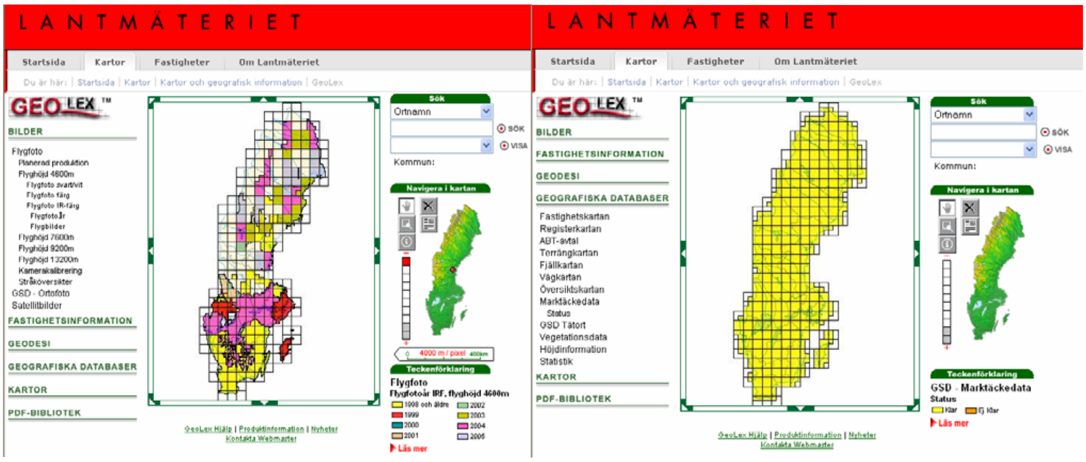

Figure 7.1 *Example of the interactive web service GeoLex available at the Lantmäteriet web site. The map to the left illustrates the age and coverage of aerial IRF photos. The map to the right shows the status of the production of the Swedish Land Cover Database, SMD. The production has been finalized, i.e. covers all of Sweden. The web service* ***GeoLex*** *is available at [http://www. geolex. lm. se/](http://www.geolex.lm.se/).*

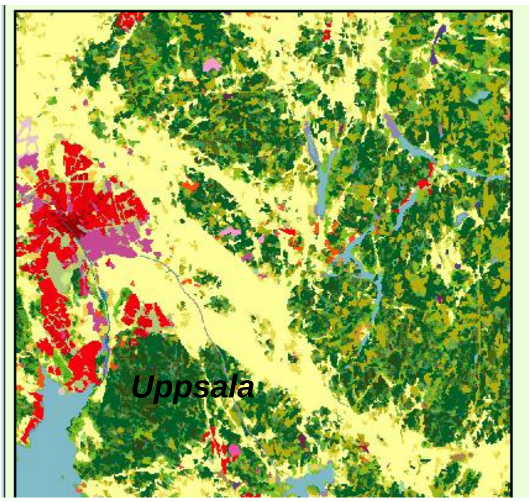

Figure 7.2 *Swedish Land Cover Data (SMD) for the topographic map 11INV over the city of Uppsala.*

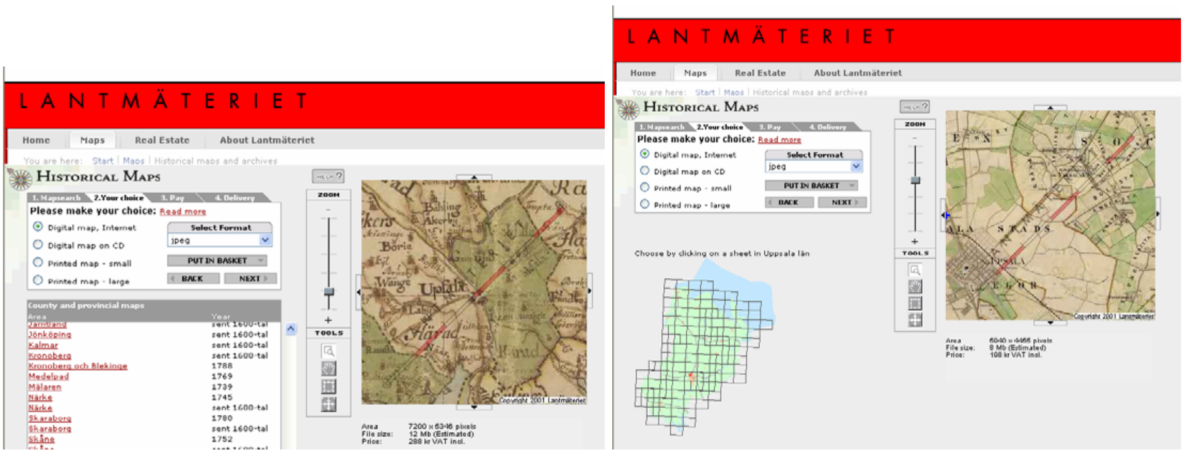

Figure 7.3 *Examples of historical maps available in digital format, from 1716 (left) and from 1859-63 (right, district map / “Häradskartan”), City of Uppsala in the centre.*

Multispectral satellite images, SPOT and Landsat, with large area coverage are also archived at Lantmäteriet. They have been procured for various projects since the 1970s, such as forest applications and the Swedish Land Cover Data production (Landsat data from ~1999 – 2001). Although the aim may have been to cover all of Sweden the same year, clouds, a short vegetation season and other factors influence the availability of suitable images on an annual basis. The mosaic of scenes for 2004 and 2005 is shown in **Figure 7.4**. The data sets archived are listed in **Table 7.1**.

Table 7.1 *Large multispectral satellite datasets covering Sweden*

Name	Time span	Spatial resolution	Ortho correction	Satellite
EPOK 1	1970s	80 m	Not complete / ok	Landsat
EPOK 2	1980s	30 m	Not complete / ok	Landsat
EPOK 3	1990s (2 cov.)	30 m	Not complete / ok	Landsat
SPOT 97	1997	20 m	Complete / ok	SPOT
SPOT 99	1999	20 m	Complete / ok	SPOT[1]
IMAGE2000[2]	1999-2001	30 m	Complete / ok	Landsat
SPOT 03	2003	20 and 30 m	Complete / ok	SPOT4 and Landsat
SPOT 04	2004	10, 20 and 30 m	Complete / ok	SPOT5, SPOT4 and Landsat
SPOT 05	2005	10, 20 and 30 m	Complete / ok	SPOT5, SPOT4 and Landsat

[1] Mainly SPOT data; [2] The IMAGE 2000 dataset has been co-processed with panchromatic Landsat data with a resolution of 15 m.

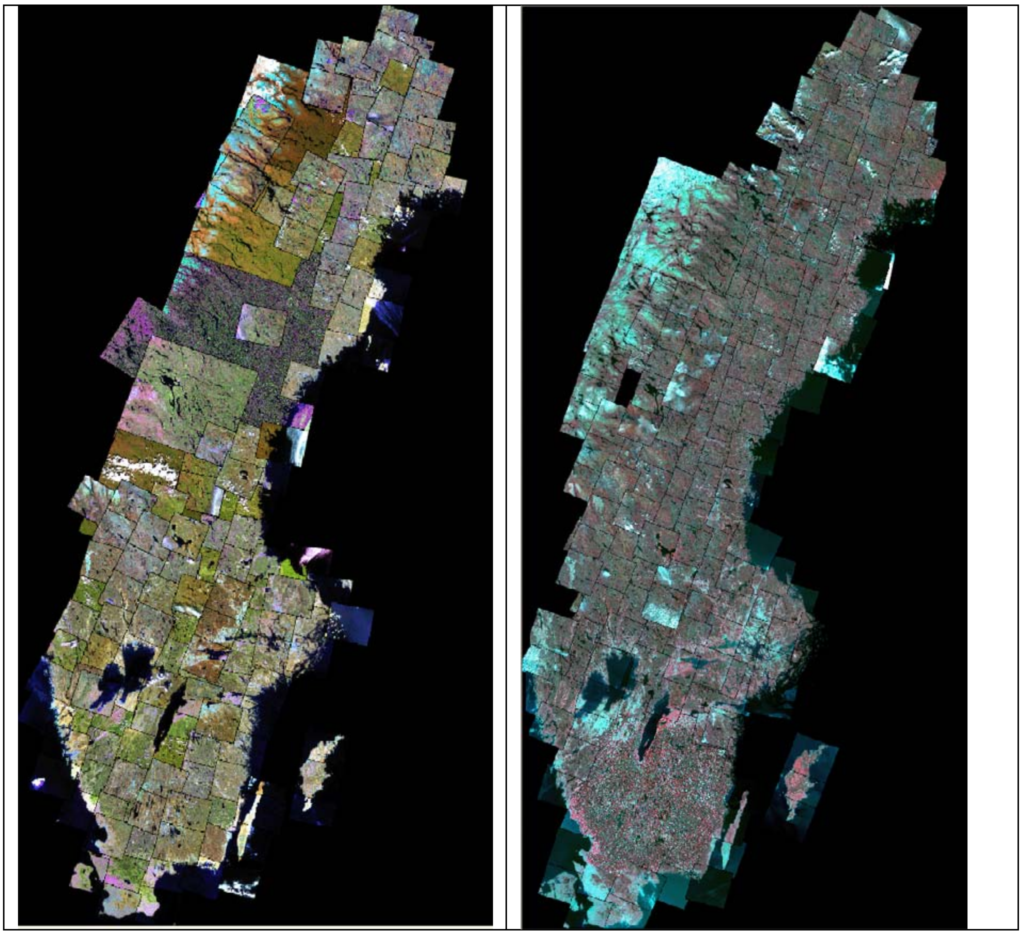

Figure 7.4 *Satellite image coverage from 2004 and 2005, SPOT and Landsat, see Table 7.1 for details.*

8 GIS and satellite data in Norway

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8.1 Introduction

In order to use GIS data for habitat modelling, we need maps that identify the critical habitat resources selected by modelled species. In Scandinavia, there has been an increase in digital map products available with national coverage during recent years. This article describes the most relevant GIS maps available in Norway for habitat modelling and their properties. Similar products are available in the other Scandinavian countries.

Norway has a limited number of sources for digital map data available at the regional and national level. These have huge differences in thematic information, and some focus more upon resource potential than actual land cover. Few maps are tested for their thematic accuracy.

8.2 Methods

Today, most digital maps are produced by operators that digitise unit borders and identify thematic classes through visual inspection of aerial photos with supplemental information from field data. Borders are determined on a digitising table from analogue maps, on screen or on 3D-workstations. Only a few satellite-based products are automatically classified through the use of algorithms.

8.3 National map products

National products are maps that are under operational production and regularly updated (**Table 8.1**). The Norwegian Mapping Authority (NMA) produces the topographic map series N50 based on aerial photos at 1:40 000 scale (www.statkart.no). The Norwegian Forest and Landscape Institute (NFLI) is responsible for detailed land-use capability maps (DMK). These data are available for areas below the tree line. NFLI is reorganising these maps into a series of new resource maps at scales from 1:5 000 to 1:2 000 000 (www.skogoglandskap.no).

Table 8.1 *National maps with regular updates.*

Map	Availability	Type	Scale	Coverage	Source	Agency
N50	Current	Vector	30 000 – 100 000	All areas	Flyfoto 1: 40 000	NMA
AR5	2007	Vector	5 000 - 20 000	Below treeline	DMK	NFLI
AR50	2007?	Vector	20 000 – 100 000	All areas	DMK, L7-sat.	NFLI
AR250	2006?	Vector	100 000 - 300 000	All areas	DMK, L7-sat.	NFLI
AR2000		Vector	300 000 – 2 000 000	All areas		NFLI
CORINE NFLI	2008	Vector	50 000 – 100 000	All areas	N50, DMK, L7-sat.	NFLI
CORINE NORUT	2007	Raster 30m	50 000 – 100 000	All areas	Landsat+N50	NORUT
DEM	2004	Raster 25m	20 000 – 100 000	All areas	N50 topograpihc	NMA
GAB	Current	Points		All areas		NMA
Vbase	Current	Lines		All areas	GPS/Aerial photo	NMA

In Norway, two national land cover products are currently being produced. Norut Informasjons-teknologi (NORUT-IT) is producing a map based on classification from clustering of Landsat data. In addition, NFLI is currently reclassifying their DMK maps according to the CORINE Land Cover classification system and with visual interpretation of Landsat imagery above the tree line. The NORUT-IT map will be available in 2007, and the CORINE map the following year. The resolution of these maps will make them suitable for regional analysis. Although these national maps are being produced simultaneously, they will probably be very different due to the methodology used and their different focus (land cover versus land production).

In addition to land cover and land use maps, a digital elevation model (DEM) covering the whole of Norway is available. These raster maps have a mean elevation reported for every 25x25 meters cell, and can easily be converted into maps of slope and aspects. GAB is a point database showing nearly all buildings in Norway and contains attributes that characterise these. All roads in Norway are coded by type and are found in a separate database called VBase.

8.4 Project based maps

Project-based maps mentioned here cover counties or smaller areas and are not regularly updated (**Table 8.2**). NFLI produces the MSFI, a raster map covering some municipalities in southeast Norway (Gjertsen & Eriksen 2004). These maps are based upon data from the point-based National Forest Inventory as a source for training data, and are produced with supervised classification of Landsat data. This resource-based map includes forestry-related parameters such as tree species composition (spruce, pine, deciduous), management classes and volume at the 30^2 m pixel level.

The NFLI vegetation map is a vector map interpreted from aerial photos and some ground truthing and is drawn by hand. Forestry inventory companies continuously produce these maps for landowners. These cover smaller areas within the suitable range of scales (1:5 000 to 1:20 000), and contain relatively detailed forestry data. However, in some areas these maps have a poor coverage.

The SatNat program, a joint program between the Norwegian Space Centre (NRS) and the Norwegian Directorate for Nature Management (DN), has examined different methods for ex-

Table 8.2 *Project related maps.*

Map	Ready	Type	Scale	Cover	Source	Actor
MSFI	Running	R. 30m	30 000–100 000	Some municipalities in SØ-Norway	Landsat+Landskog	NFLI
Forestry maps	Running	Vector	5 000 - 20 000		Flyfoto (sh eller farge)	Forest map companies
Veg. maps 1: 50 000	Running	Vector	50 – 100 000	Many places	Flyfoto	NFLI
SN Sør-Trøndelag	2006	R. 30m	30 000–100 000	ST og N-Østerdalen,	Landsat + N50	NRS/DN
SN Østfold	2004	R. 30m	30 000–100 000	Østfold	Landsat + N50	NRS/DN
SN Hardanger-vidda	2006	R. 30m	30 000–100 000	Mountain areas	Landsat + N50	NRS/DN/NINA
SN Setesdal-Ryfylkeheiene	2007	R. 10 m	10 000-60 000	Mountain areas	SPOT5 + N50	NRS/DN
Romerike	1995	Vector	10 000-60 000		SPOT3+DMK	
SN Vegetation zones	2005	R. 30m	30 000–100 000	SatNat areas	DEM25, veg. zone	

tracting data from optical satellites. Maps have been produced for the regions of Sør-Trøndelag, Østfold, Hardangervidda and Setesdal/Ryfylkeheiene. A similar map was produced in 1995 and covered 7 municipalities (2000 km²) of the Romerike plateau. The maps from SatNat are free to use and can be downloaded from DNs web site (www.dirnat.no).

Among the SatNat products are maps of the vegetation zones. These maps are similar to the vegetation zones delineated by Moen (1999), but area based upon DTEM50 and with adjustments for the effects of slope, aspect and oceanicity. The SatNat vegetation zone maps are also expressed with continuous values and not the more common categorical values. The useable scale is 1:30 000 to 1:100 000.

8.5 Accuracy

We often think that maps produced from aerial photos are reliable and very accurate. However, this is not always the case. Map accuracy is often not validated. This may be due to inherent difficulties in classifying the continuum of vegetation types into distinct classes.

A common way of evaluating map products is through an error matrix, where datasets are compared (Foody 2002). Accuracy is related to the quality of the data used to verify these classifications. Classification of vegetation can be difficult to standardize, even in field situations.

For such comparisons, map quality is expressed by parameters that describe excluding and including accuracy, total accuracy as well as statistical measurements such as Kappa (Khat) or Tau (Lillesand & Kiefer 2000). While overall accuracy is mostly affected by those classes that have a large number of samples, the Kappa and Tau measurements adjust for unevenness in sample size and thereby give a better overall comparison of estimates.

With the Kappa value it is also possible to statistically compare different maps. Very few maps have accuracy greater than 80%. Accuracy above 80% is actually very good. Common values are 0.6–0.8, but this is highly related to the classes used, and how difficult they are to identify.

In the SatNat program, comparisons were made between forestry maps and data from the point-based National Forest Land Inventory for Sør-Trøndelag county (Lieng et al. 2006). These two data sets have an overall agreement of 46% relative to tree composition. A similar comparison of tree composition from a satellite-produced map of Sør-Trøndelag (Lieng et al. 2006) with Forest Land Inventory data, gave a higher overall agreement (59%).

For mountain areas, Lieng et al. (2006) compared data from the NFLI vegetation map with point data that were sampled for the SatNat-program by the Norwegian University of Science and Technology (NTNU) Scientific Museum. The overall agreement was only 38% based on 5 broad vegetation classes. This poor consistency may be due to the spatial generalisation of the vegetation map.

Comparisons clearly indicate how different vegetation maps can be from one another when they are produced from different sources and with different methods. Therefore it is important that care is taken when habitat models are created from existing map sources.

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9 Sensors on aircraft or satellites in map production

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9.1 Introduction

Aerial or satellite images are now the dominant sources of data for map production. Satellites have long used digital sensors to measure light reflectance, and today the analogue cameras used for aerial photography have been exchanged with digital cameras. Digital cameras can capture the light reflectance from several spectral bands simultaneously, including bands outside the visual spectrum. The number of satellites delivering multispectral data is increasing, especially with regard to high resolution satellites (**Table 9.1**).

Table 9.1 *Existing and planned optical satellites suitable for vegetation classification.*

Satellite	Res. MS/PAN	# bands	Field of view MS	Year
Landsat5	30 m	6	185 km	1984
SPOT4	20/10 m	4	60 km	1996
ASTER	(15/30/90) m	(14)	70 km	1999
SPOT5	10/2.5 m	4	60 km	2002
IRS-P6 (KSS)	6 m	4	70 km	2003
CBERS-2	20/20 m	4	113 km	2003
Formosat-2 (KSS)	8/2 m	4	24 km	2004
ALOS	10 /2.5 m	4	70 km	2006
Theos	15/2 m	4	90km	-2007
CBERS-3	20/5m	4	120 km	-2008
RapidEye	6.5m	4	78 km	-2007
NPOESS (OLI)	30/15 m	8	177 km	(2009?)
IKONOS	4/1 m	4	11 km	2001
QuickBird	2. 4/0.6 m	4	16.5 km	2003
Orbview-3	4/1 m	4	8 km	2004
TopSat	5/2.5 m	3	17 km	2005
Kompsat-2 (KSS)	4/1 m	4	15 km	2006
Pleiades-1,2	2. 8/0.7 m	4	20 km	-2007
GeoEye-1	1.65/0.4 m	4	15 km	-2007
WorldView-2	2/0.5 m	8	16 km	-2008

9.2 Optical satellites

The number of spectral bands, radiometric resolution of the bands and spatial resolution determine to a large extent the quantity of information that can be extracted from the data. Coarse satellite data usually have many bands, while very high-resolution (VHR) satellites operate with only 4-5 bands. Data costs are directly related to spatial resolution, and range from being freely available to a price of approximately 200 NOK/km². The width of area covered during one pass by a Landsat satellite (30 m pixel resolution) is 185 km, while the VHR Ikonos satellite only captures a 11 km stripe.

For airborne sensors, spatial resolution is mostly determined by altitude. Digital camera images, such as with the Vexcel Ultra Cam, are based on 12-bits data and are therefore radiometrically more detailed than traditional analogue images. For images taken with digital cameras the view can be switched between natural colour (RGB) and infrared (IR) just by changing the active channel in most GIS programs, in the similar way as with multi-channel satellite data.

9.3 Classification

Since information is available on several spectra, automatic methods can be efficient for classifying images. Automatic methods for analysing image data are divided into unsupervised and supervised groups. In the first approach, image data is clustered into spectrally similar groups. These spectral groups are then interpreted into the informational classes we are looking for. Supervised classification involves identifying information classes first and using training samples to classify data into desired classes. Many statistical algorithms are available for this approach.

Most classification algorithms that are available in image analysis software are based upon parametric methods. An example is the maximum likelihood classification (MLC). However, nonparametric methods such as decision trees (**Figure 9.1**), Bayesian network and support vector machine (SVM) have recently become more common (Hand et al. 2001, Atkinson & Tate 1999). These statistical methods are not yet widely available in existing remote-sensing software, so data analysis incorporating such algorithms is more complicated.

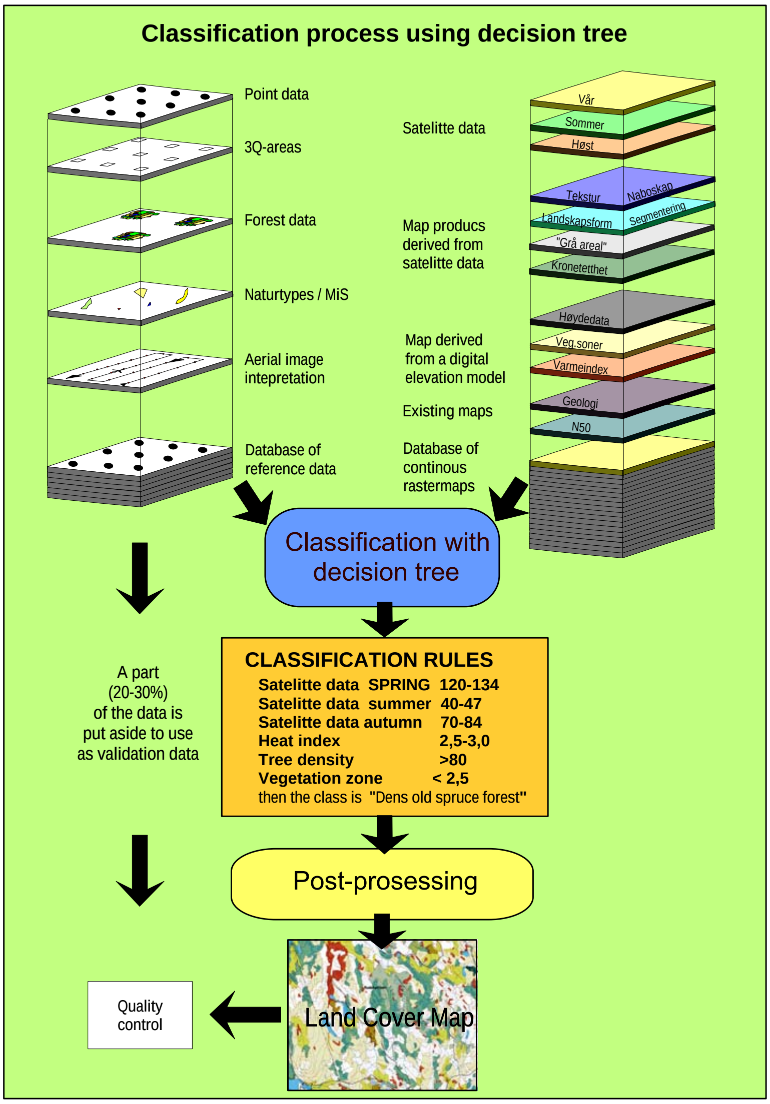

Figure 9.1 *An example of a decision tree classification process.*

When using automatic classification it is difficult to adjust for variations in light conditions within an image. Low sun angle, variable topography and wide recording angle are factors that can cause such variation. Illumination effects are therefore more pronounced in images taken with wide angle lenses such as those used for taking aerial photos as compared to the extreme telephoto lenses in satellite sensors. The height:width ratio of an image taken from an aeroplane is about 3:1, while images taken from satellites have a ratio closer to 1:40. This difference makes it difficult to use mosaics of digital aerial photos for automatic analysis. On the other hand, our brain can understand the light differences in an image, so by visual interpretation we can adjust automatically for illumination effects.

In Norway a national program was initiated in 2006 with the aim of covering the whole country with aerial photos at a pixel resolution of 50 cm during a five-year period. These data will be available through partners in the Norway Digital Cooperation, including research institutions, colleges and universities. For satellite images in Norway, there is no organised purchasing of images. In Sweden, however, several institutions are working to establish a national satellite data archive. The European Space Agency (ESA) has several programs that give researchers access to satellite data.

Images with a spatial resolution of 10 m or greater are commonly analysed at the pixel level. With high resolution data, a better approach is to segment an image into homogeneous objects before it is classified (Benz et al. 2004). One advantage of an object-based classification system is the possibility of incorporating contextual and neighbourhood information in addition to pixel values.

The SatNat program has evaluated methods for classifying VHR image data in order to train data from low resolution (LR) satellites such as Landsat. Classified products from an object-based classification are superimposed on Landsat data, and sub-pixel class coverage information can then be extracted. This method gives a better estimate of the vegetation composition within Landsat pixels than is possible from field-based registrations of training data. With better and more training data, the classification of LR satellite data will also be improved.

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10 Predicting vertebrate distribution: Database input needs for expert-generated species–habitat associations

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10.1 Introduction

It is generally accepted that species do not appear at random but reflect species specific “resource” requirements in their biotic and abiotic environments. Thus, it should be possible to predict a species distribution once its resource requirements, physiological constraints, and other limiting factors are identified. This is the basic and simple logic behind an increasing effort among ecologists to generate species-habitat associations and species distribution models, demonstrated e.g. in the National GAP Analysis Program in USA (Scott et al. 1993, 2002, Csuti & Crist 2000). The idea is, however, far from new, and ornithologists have e.g. focused on bird-vegetation associations for more than 70 years (e.g. Kendeigh 1934, Beecher 1942, Yapp 1955 1956, 1962, Knapp 1979, 1980, 1983, Bevanger 1977). An important reason for the new impetus is partly connected to improved technology (availability of satellite imaginary, the power of new computing techniques and GIS), and partly to the increased threat to species and habitats, and the immediate need for large-scale mapping of biodiversity for intelligent land-use planning (e.g. Bolstad et al. 2005).

Over the last few years several GIS-based ecological models have been developed (Gontier et al. 2007) partly because there are huge differences with respect to data availability and management needs. Model choice should of course reflect data availability, a problem neatly addressed by Wintle et al. (2005). A common situation is where presence-only (or ad hoc) data are available. In such situations, occupied locations are recorded, but no systematic attempt is made to record unoccupied locations (Wintle et al. 2005). This situation gives impetus to the application of expert- based approaches and models.

In this paper I will focus on a specific subproject being conducted as part of the Norwegian SatNat Project (Kastdalen et al. 2005, Kastdalen 2007), and describe the work procedure and discuss specific and general problems connected to the expert-based approach used. This work aimed to establish a database for a GIS-based process of predicting distribution maps of selected vertebrate species in the two Norwegian counties of Sør-Trøndelag (Central Norway) and Østfold (Southeast-Norway).

10.2 Methods

In an earlier phase of the project, a list of 165 vertebrate species (138 birds, 3 amphibians, 24 mammals) was generated. This selection was conducted by the Environmental Division of the Office of the County Governor. Moreover, distribution data from various sources, including the Norwegian Ornithological Association and the Norwegian Zoological Association, had been used to delimit each species range within 10x10 km quadrates and produce distribution maps.

The expert was initially presented a list with “resource” elements assumed to be useful in predicting species-habitat associations of wildlife by producing map layers for each resource element. After a screening and discussion process, the expert produced spreadsheets including the following elements (columns representing habitat variables and rows species):

- Norwegian vegetation zones 1:50 000 (Moen 1999);
- Digital elevation model (relating each species to possible preferences for elevation (m a.s.l.), exposition (0-360 degrees) and terrain steepness (0-90 degrees) (from N50 map data provided by the Norwegian Mapping Authority);
- N50/N250 map inclusion and exclusion element data (i.e. different environmental qualities were assumed to attract/repel a given species; roads, railways, built-up areas, water, etc.);
- Land-cover data.

Separate spreadsheets were produced for the following land-cover data:
- barren ground (vegetation lacking);
- water, snow, ice;
- built-up and cultivated areas;
- pastures/worked-up green space;
- alpine areas:
- swap/mire/wetland;
- open land and water vegetation;
- forest (> 25% tree cover).

The land cover data were assumed to be identified from LANDSAT images; however, they ended up with quite a lot more elements due to the basic units used by the project botanists. Several forest parameters such as tree species, cutting classes, wood volume per hectare and productivity class were included. Non-forested areas such as crop fields, pastures and impervious areas (e.g. roads, parking lots, buildings) with more or less than 50% green areas were also included.

The habitat variables from land cover, vegetation zones, elevation and topographic maps were coded with 0 (no association to the species), 1 (resource/habitat element associated/required by the species) or 2 (habitat type may be used by the species/insufficient knowledge of use). A separate spreadsheet was made for avoidance of (0), or attraction to (1), different environmental qualities known for the species.

After the spreadsheets were completed, GIS-experts at the Directorate for Nature Management took over the work from the expert (see Bolstad et al. 2005 for details).

10.3 Discussion

I would like to comment on two conditions related to the project – the work procedure itself and the observed weaknesses of the production of the database for the species-habitat prediction model.

The procedure of involving the expert only as a “spreadsheet producer" at a certain stage of a project should be critically evaluated. The involvement of experts as incidental subcontractors is suboptimal, since the full potential of experts is not sufficiently utilized to ensure the quality of the end-product. For instance the species selection process could be improved by involving the expert right from the beginning. The criteria used during this process may simply not meet a realistic goal.

The expert approach is basically a modelling of species distribution based on a “description of the realised niche” of a species. That is, the GIS-based ecological model is trying to "identify the niche” based on available information on coarse distribution and some known, vital species-specific environmental requirements. Unfortunately, the “niche” is a concept rather than a place, and is the sum of an organism’s tolerances and needs (cf. e.g. Begon et al. 2005). As O’Connor (2002) strikingly put it: “A GAP assumption is that one can determine a species niche accurately, that one can correctly identify the habitat or environment characteristic of a particular species". Questions and problems relating to this approach are discussed in depth by Austin (2002) and Guisan & Thuiller (2005), and I will not elaborate further on them here.

However, any expert- and niche-based modelling requires a minimum of knowledge about species-specific requirements. During this work it became evident that the available information and knowledge was insufficient regarding whether or not certain species were associated to selected and available habitat resource elements. For bird information, two standard textbooks (Haftorn 1971, Gjershaug et al. 1994) were used, and amphibians and mammals were covered by selected handbooks. However, the expert had to rely upon his own experience and subjective judgement rather extensively. Thus, the general lack of knowledge on species-specific ecology and biology are limiting factors when applying niche-based models.

Obviously, some of the future challenges will be related to predicting microhabitat preferences and associations. Unfortunately there are several species of specific environmental interest with a narrow niche, such as stenoecious species that have specific life-history requirements. However, these are not the only problems. Actually, during the last few years several critical comments have been raised regarding the shortcomings of species distribution models (e.g. Austin 2002, O'Connor 2002, Guisan & Thuiller 2005, Seoane et al. 2006), and it is obvious that significant improvements have to be made to elevate the usefulness of models if they are to become practical tools for managers within the environmental sector.

No one is questioning the general importance of the models to be able to identify habitats for species of conservation interest, and the relative importance of those variables that are imperative in influencing a species distribution. The basis of an expert-based model is the mapping of potential habitat distribution. However, if a correct mapping is made, still an important question remains – under what conditions will the species be present (cf. discussion by O'Connor 2002). For the time being, such models are static and assume unchanged environmental conditions and responses from single species to environmental variables. In other words, they only represent a snapshot. Some of the ecological, dynamic processes omitted include:

- inter- and intraspecific competition;
- predation (including human interference);
- habitat succession following disturbance;
- distribution status/phase (e.g. a species at the edge of its distribution range);
- natural population fluctuations.

In the context of global climatic change, one should be able to model habitat suitability when environmental conditions are changing. To address local or regional conservation problems as well as to handle scale problems, both high- and low-resolution predictive habitat and species distribution models have to be developed (cf. Seoane et al. 2006).

Future expert models must be more concerned with species-specific ecology and "ecological resolution". Thus, species such as the least shrew (*Sorex minutissimus*) and the brown bear (*Ursus arctos*) need to be processed differently. One should be more focused on such aspects as trophic levels (e.g. herbivores vs. carnivore species), body size, and species groups within the same and different taxa. Ecological similarities should be important factors to consider, i.e. generalist vs. specialist species, or stenoecious vs. eurioecious species.

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11 Regional habitat models – an alternative to wildlife mapping in Norway

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11.1 Introduction

Maps indicating the occurrence of animal species or suitable habitats are produced in several ways (Scott et. al. 2002). The Norwegian Directorate for Nature Management began early to produce maps that showed important habitat for different wildlife species (DN 1996). Those maps were made mostly from knowledge of local people. These informant-based maps were difficult and costly to keep updated and suffered from a lack of objectivity. Maps were imprecise due to rapid landscape changes or the lack of informants in some places.

Alternative methods are used to produce wildlife maps through the use of habitat models that are based either upon empirical observational data or expert knowledge of species habitat associations, or a combination of the two. Observational models are linked to the data that derived them, and it is often difficult to sample observational data representative for large areas. Good empirical data can be used to produce precise maps. Expert-based models are more general, are easier to adapt to new areas and have been shown to work well for large areas like counties.

11.2 The expert model

As part of a joint program (SatNat) between the Norwegian Directorate for Nature Management (DN) and the Norwegian Space Centre (NRS), a production line for creating county-level maps of potential habitats has been developed (Bolstad et. al. 2005). The production line generally follows the procedure used in the Gap Analysis Program of the United States (Csuti & Crist 2000), with adaptations to available data sources in Norway. The process of making maps has been automated through the development of an extension to ArcView 3.x. This article describes the analysing process in this extension. The production of a database with habitat association for a selection of wildlife species is described by Bevanger (2007) and more detailed information of the satellite-based land cover data is described in Vikhamar et al. 2004 and Lieng et al. 2006.

Before making the habitat models, it was necessary to pre-process the data. Distribution data from the Norwegian Ornithological Association and the Norwegian Zoological Association (ATLAS data) together with species distribution data from the DN Naturbase are combined and converted to raster layers. For 168 species maps are produced showing those quadrants of 10 x 10 km where the species have been observed.

In order to map the habitat parameters, we used a standard topographic map in scale 1:50 000 (N50), a map of vegetation geographical zones (continuous data) in the same scale (Lieng et al. 2006), a digital elevation model with 25 meter pixel resolution and a land cover map derived from Landsat satellite data with 30m meter pixel resolution.

***Table 11.1** The different landscape parameters used in the model for Sør-Trøndelag county.*

Main groups	# of classes	Classes used in the model
Open areas at lower elevation	10	Sparsely vegetated, lichen-dominated, heath/ shrubland, meadows, other grass-dominated areas, coastal heathland, developed areas, shoreline, seashore meadows
Agriculture areas	3	Fertilized and surface treated, fertilized pasture, grazing land
Bogs	3	Bog with trees, wet bog, open bog
Forest	11	Clearcut, young coniferous forest, dense spruce, open spruce, dense pine, open pine, sparsely vegetated, unproductive land, broadleaved forest, dense boreal forest, open boreal forest, mountain birch forest
Mountains	6	Barren areas, sparsely vegetated areas, lichen-dominated areas, heath/shrubland, meadows, other grass-dominated areas
DEM	3	Elevation, slope, aspect
N50	8	Sea, lake/pond, river/stream, glacier, wetland, road, railroad, developed areas

The pre-processing include dividing the data into several thematic layers and convert the vector data to raster formats equivalent to 30m Landsat pixels, reclassifying the vegetation zones into standard categorical zones, making maps of slope and aspect and separating the land cover data into 5 broad groups with 3-11 classes each (**Table 11.1**).

Habitat associations were a central part of the model. We created a database of habitat associations in spreadsheet format, with columns representing habitat variables and rows representing species (Bevanger 2007). These were derived from expert knowledge and literature studies. We classified each map category (**Table 11.1**) as either “used” or “not used” by the species, or “unknown” when we did not have sufficient data or knowledge. We identified values for upper and lower limits for maps with continuous variables.

A given wildlife species tends to prefer or avoid different features of the environment. We coded such data (roads, water, built-up areas, etc.) as “including” or “excluding” and identified an effective buffer distance. Buffer distances were only used when avoidance of, or attraction to, different environmental qualities were known for the species.

The Arcview 3.x extension conducts analysis for a species selected by the operator. A Microsoft Access database is linked to the extension by ODBC, and the extension makes complicated calculations which link the information from the habitat association database with all the different raster grids.

In a post-processing analysis, the map of potential habitats is spatially generalised by using a majority filter, and the distribution delimited by the 10 x 10 km quadrant map of observational data. As a result two maps are produced, with one map showing potential habitat and the other the probable species distribution. The species distribution is only a limitation of the potential habitat based upon the result from the 10 x 10 km quadrant map of observational data. It is possible to save temporary maps made during the calculations. These maps are useful for identifying where the models can be improved.

For Sør-Trøndelag county (19 000 km²), the analysis for one species takes approximately 3 minutes on an ordinary desktop computer. A manual analysis of the same data would take several days. **Figure 11.1** shows an example from the model for two species in Sør-Trøndelag.

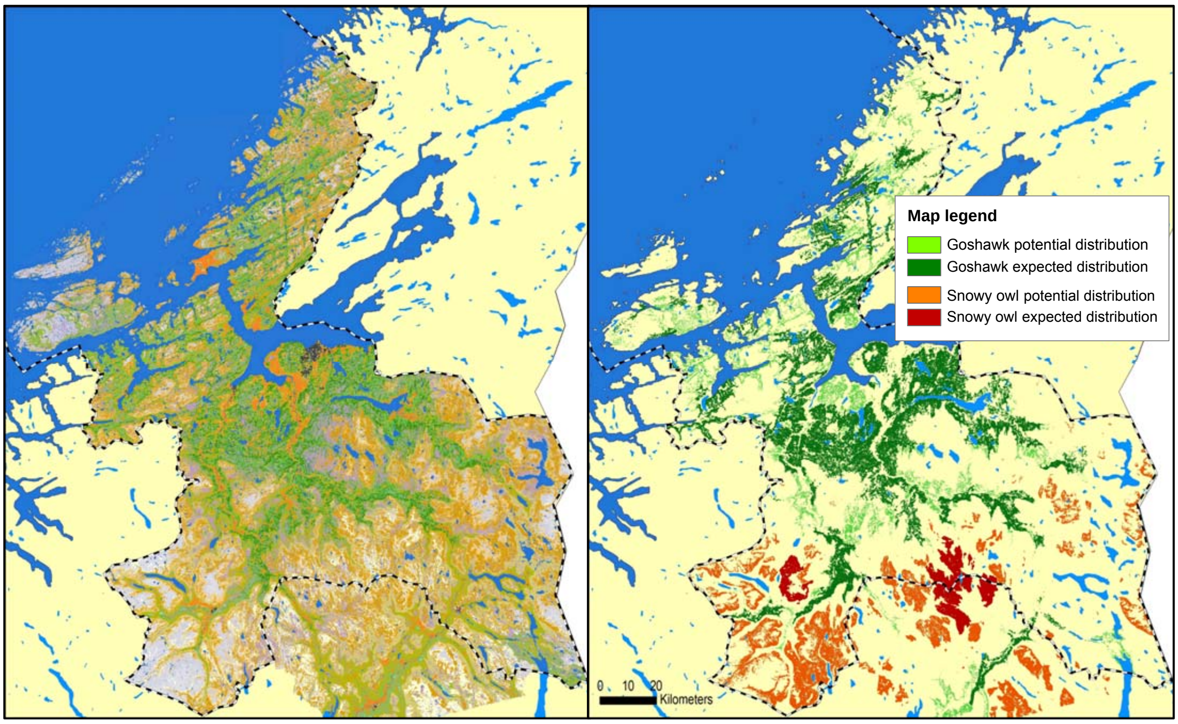

Figure 11.1 *Model predictions for Goshawk and Snowy Owl in Sør-Trøndelag and northern part of Østerdalen. Predicted is before using observations within the 10 x10 km atlas quadrants to adjust the distribution, expected is after adjustment.*

11.3 Validation

So far the results have not been comprehensively validated. Few good data exist for conducting an accuracy assessment. In Østfold county, Finne (2005) compared the distribution maps from an expert model with observational data for four avian species. Data from Nightjar (*Caprimulgus europaeus*) and Woodlark (*Lullula arborea*) were sampled by drawing circles with a radius of 100 meter around those areas were the species had been observed. Counts of Black Grouse (*Tetrao tetrix*) and Capercaillie (*Tetrao urogallus*) were conducted by hunters with free-ranging pointer dogs in early August. Both the bird observations and the track of the hunter were determined with GPS. A 60 m buffer around the tracks was defined as the study area. Within the defined study area the percent of estimated habitats and the number of observations in estimated habitats were calculated. Finne suggested that the habitat model worked well for predicting a distribution if there were a low percentage of estimated habitats inside the sampling areas and the observational points covered a high percent of suitable habitats estimated by the model.

There were good matches between the habitat model and the observations for Nightjar and Woodlark (**Table 11.2**). The habitat associations used in the model are clearly capturing important habitat variables for these two species. Capercaillie had a high percent of observational points in estimated habitat (92%), but also a high percent of estimated habitat inside the buffer. Black Grouse had a low percent (43%) of estimated habitats in controlled areas and a similar proportion of observational points in estimated habitat (45%). The result for the Capercaillie is not unexpected due to the high percent of estimated habitat inside the controlled area (the buffer area). A conclusion from the assessment of Capercaillie and Black Grouse was that the habitat variables in the land cover map were not detailed enough to give a good identification of important summer areas.

***Table 11.2** Accuracy assessment based on observational data and suitable habitats estimated by the model for four forest-dwelling species.*

Species	percent cover of estimated habitats in controlled areas	percent cover of observational points in estimated habitats
Nightjar	39	85
Woodlark	37	76
Capercaillie	84	92
Black grouse	43	45
Capercaillie leks	39	77

For capercaillie, 49 leks were located with GPS in the eastern part of Halden municipality. Nearly 80% of the leks were located in habitats estimated from the model, while the study area cover less than 40% of these preferred habitats (**Table 11.2**). These results indicate that the estimated habitat map gives a good prediction of the distribution of potential lek areas.

Recommendations from the accuracy assessment on four species indicate that the present habitat model is useful as a supplement to wildlife mapping based on field surveys to delimit areas of interest. This is especially valid for species with a good match between the habitat model and the land cover variables.

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12 Model adaptation to road planning

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12.1 Introduction

Two different kinds of models were used in tests for predicting wildlife distribution and movements on a landscape scale. One was developed from an animal resource availability perspective, where ring-neck pheasant (*Phasianus colchicus*) was used in the test studies. The other was a random walk model used for hedgehog (*Erinaceus europaeus*) and moose (*Alces alces*).

12.2 The original resource availability and pheasant distribution model

Data and experience obtained from research on pheasant ecology and ethology in the 1970s and 1980s in southern Sweden were used to develop a simple model based on the availability of required essential resources available to single individual birds. The model was kept as simple as possible. Only four different parameters were used: (1) cover from nocturnal predatory mammals like the red fox; (2) cover from diurnal raptors such as the goshawk; (3) areas suitable for foraging by adult pheasants and (4) areas suitable as pheasant breeding grounds. A background map showing the distribution of different habitats providing essential resources was obtained by using a variety of the European CORINE land cover data developed for Swedish conditions (Svenska marktäckedata, SMD) (Ahlcrona 2002).

The model was able to select land cover classes from the SMD layer appropriate to the specific needs of resources important for pheasant survival. Trees provided cover from the red fox, since pheasants roost in trees at night. Thus forests of different kinds, as well as smaller stands of trees, were assigned as fox resistant. Thickets and brushwood, along with different stands of bushes, were expected to provide shelter from goshawks. Most foraging activities in pheasants occur in rather open land covered with bare ground or field layer vegetation. Thus, arable fields, grasslands and other kinds of open land are classified in this model as foraging grounds if they are near avian predation cover. Suitable pheasant breeding sites are covered with dense field layer vegetation that is high enough to protect a ground nest from visual detection and assure a moist and stable micro-climate.

The model selects patches from the SMD layer representing the different ground coverage classes that meet the four prerequisites of a single pheasant territory mentioned above. Circular areas of 200 meter radius (area = 12.6 hectares), which represented mean territory size, were freely moved over the SMD layer in order to find sites where all four habitat types were present within circles. New circles were moved and fitted by encircling the four necessary habitat classes and finally fixed.

In this way, potential non-overlapping pheasant territories are distributed over the landscape and occupy possible territory locations. The resulting map shows the geographical distribution of pheasant territories as defined from the habitat parameters included. In this simple model it sufficed if the four habitat classes were represented within the same circle unless they represented larger or smaller fractions of the territories.

The distribution of potential territories in a landscape can be visualized by using a neighbour analysis. If the simulated territories are symbolized by points it is possible to construct Dirichlet tessellations or Thiessen polygons around them (Wray et al. 1992). In this process, straight lines will connect a certain point with its neighbouring points. These lines intersect at the middle by right angular lines, which cross each other and inscribe the point within a polygon – a Thiessen polygon. The shorter the distances are to neighbours the smaller the polygons will grow and *vice versa*. Thus, areas with a higher density of territories will show a pattern of smaller polygons. If small polygons are represented with dark colours and larger polygons with light colours, core distribution areas will be dark in colour.

12.3 An improved pheasant resource availability model for evaluation of nature quality

An improved model was developed (Nordström 2003) where specified minimum amounts of different essential resources had to be available. A nature quality index was then calculated in accordance to the amount and the combinations of occurring resources (1) from a SMD layer in a GIS environment.

An Avenue script was developed (Nordström 2003) using the formula (1) for a circular window (radius 200 m) which is systematically moved over the SMD layer pixel by pixel and row by row. For every position a ($I_{pheas.}$) value is calculated and recorded in a map according to the position of the centre pixel. After the calculations are completed a map is generated that shows areas with different ($I_{pheas.}$) values which can be highlighted with colour coding.

Requirements

Parameter requirements	Ground cover and (SMD-code)
GPC. Ground predator cover	Coniferous forest (312121, 312122) Mixed forest (3131)
B. Number of GPC-areas within the circular area	-
RPC. Cover against raptors	Bushes, brushwood (3241) Deciduous forest (3111) Young wood (32431, 32432)
BG. Breeding ground	Deciduous forest (3111) Natural grassland (321) Young deciduous forest (32432) Regenerating clearcut (32422) Limnogen wetland (411) Brushwood (3241) Arable land (211) Shores (51) Marsh/Tall herb meadow
FG. Feeding ground, variation (for arable land)	Arable land (211) Grazed meadows (231) Deciduous forest (3111) Natural grassland (321)
F. Number of FG-areas if arable land (211). If F >4 => F=4	-
FGC. Feeding ground for small chicks	Grazed meadows (231) Natural grassland (321) Moor land (322) Sparsely vegetated areas (333)

Adjustable constants

K0 = radius (default 200 m)
K1 = default 0,1
K2 = default 0,02
K3 = default 0,1
K4 = default 0,02
K5 = default 0,02
K6 = default 12,6

Quality index $_{\text{aktivity area, pheasant}}$ = ($I_{pheas.}$)

$$(I_{pheas.}) = [A(1+(K1*B))+(C+D)(1+(K2*(L_{CE}+L_{CG})))+E(1+(K3*F)+(K4*L_{CE}))+G(1+(K5*L_{CG}))]/K6 \quad (1)$$

A Σ GPC-areas > 0,02 ha;
B Number of GPC-areas within 100m distance from another GPC-area
C Σ RPC-areas > 1 ha; advantageous to face FG (L_{CE}) och F (L_{CG})
D Σ BG-areas > 0,1 ha; advantageous to face FG (L_{CE}) och F (L_{CG})
E Σ FG-areas > 2 ha; advantageous to face RPC (L_{CE})
F Number of FG-areas if they are arable land (211) (number not exceeding 4)
G Σ FGC-areas >2 ha; advantageous to face C (L_{CG})

Such models have also been developed for other species, including the grey partridge (*Perdix perdix*), the nuthatch (*Sitta europaea*), the European hare (*Lepus europaeus*) and the badger (*Meles meles*). Models will be validated using data collected by territory surveys and radio telemetry.

12.4 Random walk models for hedgehog and moose and a static one for roe deer

The random walk model used for the hedgehog is based on a stepwise movement generator where the step-length is variable and depending on what habitat class the movement is passing - longer steps and faster movement in non-attractive habitat classes than in attractive ones (Göransson 2004). Other movement models are “least-cost” models which analyse resistances in the landscape that will create energetic costs to overcome (Adriaensen et al. 2003).

Furthermore, the step direction is randomized, but completed by a memory function for good locations and through sensitivity for positive/negative changes in habitat attractiveness. The tendency for avoiding backward directed steps was also adjusted for. Movement tracks were simulated on a habitat map classified from a high resolution infrared (IR) aerial photo. These simulations displayed movement patterns similar to those obtained from radio-tracked hedgehogs.

For moose, a random walk model was also used in order to simulate movement tracks in a heterogeneous forested landscape. This analysis was conducted in order to identify places in the landscape where moose can cross roads and thus pose threats to traffic safety. Similar to the hedgehog model, the moose model was sensitive to habitat distribution obtained from the SMD layer. However, movement was also influenced by topographical data in order to minimize energy consumption.

This model will be validated along roads where moose movements are recorded by snow tracking and by faecal pellet counts. A further possible way to validate such a model is by comparing actual moose movements from telemetry data with computer-simulated movements in the same landscape.

Finally, the static model used for roe deer (*Capreolus capreolus*) was applied in an agricultural landscape in order to predict where roe deer were expected to cross roads. A regular pattern of points was created over a road map in lines parallel to the road and at different distances from the road. Around these points buffer areas were calculated and used in order to clip a background map classified for suitable habitats. For each buffer area, a quality index similar to that of the pheasant model was calculated. If not all of the needed habitats were included in a buffer area that buffer was excluded.

As a result, a graded map was obtained showing the expected probability of roe deer distribution in the landscape close to roads. These predictions were compared with real instances of roe deer – vehicle accidents and were evaluated as being realistic.

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13 Maintaining biodiversity – gap analysis and landscape planning with GIS as tools

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13.1 Introduction

Conservation of biodiversity is a major challenge to achieve sustainable development. A fundamental starting point for success is that stakeholders at the policy level, within the scientific community, and among actors involved in governance and practical management share the same understanding of what sustainable landscape management means. This involves both terrestrial and aquatic ecosystems (Törnblom 2007). How society's different institutions and actors can achieve biodiversity conservation by inter-sectoral collaboration is thus a key issue (Birot et al. 2005).

"Biodiversity" is a complex term that encompasses the composition (genes and species), structure (habitats) and function (processes) of ecosystems. Successful conservation of biodiversity can be defined as all naturally occurring ***species*** with populations existing in viable populations found in representative and functional stable or dynamic ***habitat networks*** that are maintained by ecosystem ***processes*** at multiple spatial and temporal scales. The extent to which forest landscape biodiversity is maintained is thus a matter of levels of ambition in space and time:

1. species may be present, but not in viable populations:
2. viable populations may be present, but only those that are not specialised on natural forest structures or do not have large area requirements;
3. communities of all naturally occurring species of the representative ecosystems of an ecoregion are present and ecosystem services are functional, but external impacts from the surrounding, large scale disturbances and global change can threaten ecological integrity;
4. resilient social-ecological systems are in place with adaptive ecosystems and governance systems.

Usually regions with a long history of large-scale transformation of natural or cultural landscapes have an impoverished status of different dimensions of biodiversity than those having a short history. To implement sustainable development, there is a need to evaluate the extent to which different biodiversity conservation tools such as forest management practices and different measures such as forest certification, protected areas, or national parks, actually will maintain biodiversity, and with what level of ambition.

Such evaluations require systematic approaches to make assessments of the status and trends of different elements of biodiversity. Assessments should be made such that the results can be communicated among the actors affecting the status and trends of biodiversity. Because in most cases habitat loss is the most important threat to biodiversity, analyses should focus on analyses of the functionality of habitat networks at multiple spatial and temporal scales and over long time.

13.2 A landscape approach to landscape assessment, planning and management

The appearance of the global sustainability discourse is associated with a wide range of policy documents that advocate, in one way or another, a landscape approach where different economic sectors are integrated within a geographic area. The landscape concept implies, however, several things. One important dimension is to include a wide range of spatial scales, from points and patches such as fields or forest stands, to landscapes, administrative units and regions. Another dimension is the need for interdisciplinary approaches to management that require input from natural as well as social sciences, and the humanities. A final example is the call for encompassing a wider range of themes or information layers that include both tangible and non-tangible values. A landscape approach to the assessment of biodiversity for planning and management of for example a forest management unit or a municipality thus needs to focus both on the ecosystems, and the institutions and actors involved with planning and management (Angelstam et al. 2003a, b; Lazdinis & Angelstam 2004, Angelstam & Elbakidze 2006, 2007). This approach involves studies of both ecological and institutional dimensions using both quantitative and qualitative methods.

13.3 Mapping ecosystems, gap analysis, and landscape planning

As a base for reaching these different levels of ambitions, mapping of ecosystems at multiple spatial scales is needed. To make spatially explicit analyses, digital data regarding the quality, size, connectivity and matrix surrounding (e.g., forest, mire complexes, tundra, or agricultural land etc.) are necessary. With the mapped areas with high conservation value as an estimate of the assets for functional habitat networks, one should then determine what level of ambition of biodiversity maintenance is desired. Tools for systematic biodiversity assessments are available for systematic conservation planning for the maintenance of biodiversity at strategic, tactical and operational levels.

At the strategic level, one can both analyse the representation of different terrestrial and aquatic ecosystems using gap analysis. This involves comparisons between the present and desired amount of different land cover types needed to maintain natural or cultural biodiversity. At the next step, the tactical level, the quality of habitat networks is evaluated. Here all stakeholders should be made aware that there may be thresholds for habitat loss, which if exceeded, will lead to loss of biodiversity. Because changes in the viability of populations are difficult to detect, the process of biodiversity degradation is often underestimated. Finally, at the operational level landscape planning to realise this ambition by combining protection, management and restoration of forest ecosystems and processes at multiple spatial and temporal scales is made with relevant actors. This approach can be made for both terrestrial and aquatic ecosystems (Degerman et al. 2004).

Planning level	***Purpose of planning***
Strategic	Provide a policy level message about the state of biodiversity within each ecoregion
Tactical	Provide spatially explicit maps indicating the functionality of terrestrial and aquatic infrastructures of landscapes
Operational	Give management recommendations to actors (e.g., land owners, for management units and administrative districts)

13.4 Mapping institutions, and actors' understanding, ability to act and attitudes

To conduct integrated analyses of ecological sustainability of landscapes, research methods from both the natural and human sciences are needed. An important part of the social dimension of such an analysis involves studying the actors and institutions implementing policies in a landscape or region, as well as the affected actors and institutions – roles that do not necessarily overlap. This would include the identification of the actors and mapping of networks, the mapping and investigation of issues of concern at different levels and for different actors, and an evaluation of the implementation process as a whole. This evaluation would concern how policy is formulated, how policies are translated into regulations, and how these are communicated and implemented in a defined social-ecological system.

However, such an evaluation would also look at the extent to which policies and regulations are allowed to learn from, and thus be reformulated by, local experience of actual implementation. With this approach the actual (and potential) feedback loops between policy, implementation and effects on the ground, or alternatively the lack of feedback can be understood, and improvements can be identified and suggested (Elbakidze & Angelstam in press).

Mapping institutions
Planning, management and user activities that affect landscapes take place at multiple spatial scales. To understand the implementation structures, a fundamental step is to identify the formal and informal institutions that work with planning and management (directly and indirectly) of different kinds. These could include clearly directed efforts, such as those performed by institutions involved with forestry, road networks, drainage systems, agriculture, conservation areas, mining, and oil/gas, but also work and visions that more indirectly affect planning and management, such as academic education, vocational training and media. The mapping of so-called implementation structures should start from the ”bottom” (e.g., from managers and individual land owners) in actual landscapes and reach the regional, national and international arenas and their different actors and interest groups.

Understanding implementation processes
The implementation of environmental sustainability goals requires the collaboration of many actors who often have different interests and agendas. Here one needs to analyse the guiding policy documents: How consistent are policy goals and instruments between different policy areas, sectors and levels of governance? Are important aspects relevant at “bottom” levels, like compositional, structural or functional elements of ecological and socio-cultural sustainability, left out on the regional, national or international arenas? To what extent are they actively integrated?

13.5 Examples of projects applying systematic landscape assessment

<u>Whole Sweden 1996-97, 2006-07.</u> Upon request from the Swedish government an analysis was made of the amount of forest needed to be set aside to conserve viable populations of species in Sweden. Scientific report in Angelstam & Andersson (2001). The work was revised for the Swedish Forest Agency in 2007 (Angelstam et al. 2007).

<u>The counties Dalarna and Gävleborg 2002-03.</u> As a follow-up to the national level gap analyses two counties (55 000 sq. km) commissioned an eco-regional gap analysis and spatially explicit mapping of high conservation value forests using complex GIS analyses (Angelstam et al. 2003).

Estonia 2002-03. Based on the same approach as for Sweden an analysis was made for Estonia regarding how much forest that ought to be set aside for the conservation of viable populations (Lõhmus et al. 2004).

Latvia 2003-05. As a follow-up to the nation-wide analysis of high conservation value forests (woodland key biotopes) in Latvia, the forest service and state forest company commissioned a nation-wide gap analysis (Angelstam et al. 2006).

Lithuania 2007. Analyses of forest habitat representativity for all of Lithuania was made by Angelstam & Mozgeris (in press).

Carpathian Mountains, Skole region, Ukraine 2005. Within the framework of an EU COST action on sustainable forest management an analysis was made regarding the institutional setup to implement different sustainability dimensions, including biodiversity (Elbakidze & Angelstam 2006).

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14 Resource selection functions

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14.1 Introduction

Maps of suitable habitat are based on the actual selection of a habitat type by individuals or by groups. Selection of a habitat with certain resources, in this context, will be the process of an individual actually selecting one or more resources within this habitat (Johnson 1980). Selection can be divided into several orders. A first order selection is the selection of the range of a species. A second order selection is the selection of a home range within the total range of the species. A third order selection, in which we will focus on, is the selection of resources within a habitat. At last, the fourth order selection considers the selection of types of foods among those that are available to the species (Johnson 1980).

One method to approach habitat selection is through Resource Selection Functions (RSF's: Alldredge et al. 1998, Boyce et al. 2002). A RSF is a mathematical and statistical function based on our knowledge of animal locations and their overall habitat demand. A RSF can be defined as any function that is proportional to the probability of use by an organism. Habitat selection can deal with variables on either continuous or discrete or both levels. Discrete variables can be categories of land cover such as wet bogs, deciduous forest, or coniferous forest. Continuous variables include distance to water, distance to roads, aspect, or slope (Manly et al. 2002). RSF's are efficient tools for species management when they are combined with observational data of animals and habitat information from satellite data or other sources in a GIS (Boyce & McDonald 1999).

An RSF is a kind of habitat suitability index (HSI), but differs from HSI with regard to input data. While HSI can be based on empirical data as well as expert opinions, RSF's are strictly estimated with empirical data from observations of presence/absence (used vs. unused) or presence/available (used vs. available) resource units (Boyce et al. 2002). Common data sources include relocations of marked individuals (Moa et al. 2006), dens, nests, carcasses or other tracks (LaHaye & Gutiérrez 1999, Andersen et al. 2005a,b). However, any observation of a species can be used, for example the locations of individual plant species.

The most common statistical tool for handling data of used/unused or used/available is generalized linear models (McCullagh & Nelder 1989). Several methods are developed to validate these types of models (Burnham & Anderson 2002, Venables & Ripley 2002). Validation is important for the predictive value and for the reliability of these models as resource management tools (Boyce et al. 2002).

Although RSF models are robust and widely applicable, there are statistical issues that might be of concern when using them. This relates in particular to the lack of handling of spatial and temporal autocorrelation by RSF, and the creation of random points to represent available units (Lennon 1999). Thus, caution should be taken regarding these matters when evaluating the resource selection functions. Most statistical software includes tools for checking of autocorrelation.

14.2 Studies with RSF's implemented

An RSF model can be applied to studies where both time and scale are issues (Manly et al. 2002). Selection studies where time is an issue are often motivated by providing information on the long-term resource requirements of a species (Laymon et al. 1985, Schoen & Kirchhoff 1985), and by the fact that the habitat selection often changes over time, e.g. due to different availability of resources between seasons (Moa et al. 2006). Several studies use resource selection functions at multiple scales to map the distribution of different species (Erickson et al. 1998, Johnson et al. 2004, Kastdalen et al. 2003, Kastdalen & Gundersen 2004). Multiple scales are used to account for a species' perception range. Resources may be selected at various scales. In addition, habitat models at multiple scales may serve different tasks in management of a species.

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15 Can deer-vehicle collisions be predicted?

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15.1 Introduction

Collisions between deer and motor vehicles (DVC) are considered as an important threat to traffic safety, socio-economics, animal welfare, wildlife management and conservation in many countries worldwide (Groot-Bruinderink & Hazebroek 1996, Seiler 2004). In Sweden, DVC have multiplied over the past thirty years, with up to 5 000 accident reports involving moose (*Alces alces*) and 25 000 reports on roe deer (*Capreolus capreolus*) annually (Seiler 2004). The actual number of collisions with these animals is likely twice as high (Seiler et al. 2004). Besides the obvious traffic safety aspect, there are economic, ethical and ecological motives for reducing DVC.

Various measures to mitigate DVC have been tested in Sweden, but only exclusion fencing and roadside clearing have proven efficient (e.g. Anonymous 1980, Björnstig et al. 1986, Skölving 1985). Can the risk for DVC be foreseen? DVC are non-random, temporally and spatially aggregated (Seiler 2005). If high-risk periods or road sections can be predicted, mitigation measures may be chosen and located more efficiently and adverse effects on wildlife be minimized. I present studies on spatial and temporal patterns in DVC in Sweden and discuss predictive models on moose-vehicle collisions (Seiler 2004, 2005).

15.2 An overview

Factors responsible for the occurrence of animal-vehicle collisions can be summarized under three major categories: (a) the animal, its ecology and behavior, (b) the traffic, its density and velocity, and (c) the environment, including the road as well as the surrounding landscape. The interplay of these factors creates a complex pattern in the spatial and temporal distribution of animal-vehicle collisions that must be understood before effective countermeasures can be designed and employed. Spatial patterns may reflect local variations in animal abundance and activity, habitat distribution, landscape topography, and road and traffic characteristics (e.g., Groot-Bruinderink & Hazebroek 1996, Hubbard et al. 2000, Nielsen et al. 2003, Clevenger et al. 2003). Temporal patterns may be caused by seasonal and diurnal variations in traffic volume, weather and light conditions, and animal activity associated with, e.g., foraging, mating or breeding behavior (e.g., Reh & Seitz 1990, Jaren et al. 1991, Gundersen et al. 1998, Mysterud 2004).

Long term trends in DVC in Sweden (at national and regional scale) are highly correlated with inter-annual changes in deer harvest and the steadily increasing traffic volume (Seiler 2004). Naturally, the number of animal-vehicle collisions should be a function of the density and activity of animals and vehicles. However, with increasing spatial and temporal resolution, DVC patterns become more diffuse and other factors such as behavioral, environmental and traffic flow factors gain significance. Ultimately, it is the behavior of the animal in response to the vehicle and the reaction of the driver on the approaching animal that determine whether an accident happens or not. Understanding these factors will give the key to predict future collision risks and develop efficient prevention.

15.3 Modelling deer-vehicle collisions in Sweden

Various models on DVC have been developed over the past years (e.g., Bashore et al. 1985, Child 1998, Finder et al. 1999, Hubbard et al. 2000, Nielsen et al. 2003). For such models to be applied in road management, however, it often requires a greater simplicity and generality than what the models allow. In addition, parameters should be chosen not only for their predictive power but also with respect to whether they can be controlled by the road planner, the driver or the landowner.

In my studies (Seiler 2004, 2005), I developed logistic regression models to predict the risk for vehicle collisions with moose on public roads based on remotely sensed landscape data, road and traffic statistics, moose harvest (as an index of population densities), and collision records from 1990 to 1999. I quantified environmental data from 2000 accident and 2000 non-accident control sites (located at least 1 km away from any reported accident site) in south-central Sweden (mainly the county of Östergötland) and used multiple logistic regression analyses (Hosmer & Lemeshow 1989) to identify which of the 25 road, traffic, and environmental parameters that significantly distinguished between observed accident and non-accident control sites.

Site status (accident or control) constituted the binary response variable that I used in the following logistic regression analyses. These were based on three a priori models representing parameter combinations useful in different phases of road management. The simplest model ('road-traffic model') included only basic road and traffic parameters that are readily available in strategic environmental impact assessment (SEA) or strategic road management. The second model ('landscape model') contained only parameters obtained from remotely sensed landscape data and digital maps providing information on the 'background' DVC risk in an unexploited area. Also this model could be used in SEA. Once the routing of a new road is decided, or if the road in question already exists, data on road, traffic and environment can be combined with additional information on the juxtaposition of landscape elements relative to the road. The 'combined model' thus provides a tool for the evaluation of alternative routes in environmental impact assessment (EIA) at project level or for the identification of road sections with high risks of DVC.

For the construction of the multiple models, stepwise (backward) regression procedures were used to identify significant parameter combinations. The different variable sets were then compared using Akaike's Information Criteria (AIC) and Akaike weights (w_i) to identify the most parsimonious model (Burnham & Anderson 2002). Model structure was considered adequate if variance inflation factors (ratio of goodness-of-fit χ^2 statistic to its degrees of freedom) were close to 1.0 (Cox & Snell 1989, cited in Burnham & Anderson 2002).

The predictive abilities of the subsets for the three a priori models were tested in the county of Örebro on 2600 regularly distributed 1-km road sections classified as either accident sites (if at least one moose-vehicle collision (MVC) occurred during the period of ten years) or non-accident sites (if the distance to the nearest MVC location exceeded 1 km). As during model construction, environmental parameters were quantified within 500 m radius around the centrepoint of each accident or control road section. To determine how the different models performed in distinguishing between accident and non-accident sites in the test area, univariate logistic regression analyses were used with the actual status of the test sites as binary response variable and the site specific probability for MVC as independent predictor.

Traffic volume, vehicle speed, and the occurrence of exclusion fences appeared as the dominant road-traffic factors determining collision risks, identifying 72.7% of all accident sites. Within a given road category, however, the amount of and distance to forest cover, density of intersections between forest edges, private roads and the main accident road, and moose abundance indexed by harvest statistics significantly distinguished between accident and control sites. In combination, road-traffic and landscape parameters produced an overall concordance in 83.6% of the predicted sites and identified 76.4% of all test road sections correctly.

15.4 Conclusions

The risk of moose-vehicle collisions in Sweden can be predicted from remotely sensed landscape data in combination with road traffic data. Prediction models suggest that reduced vehicle speed in combination with road fencing and increased roadside clearance may provide effective tools to road planners in counteracting MVC. However, effective mitigation will depend on integrated management of the surrounding landscape and of the moose population, as well as increased responsibility by individual drivers. Remedying animal-vehicle collisions must involve road authorities as much as landowners and the public.

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16 Mahalanobis distance and poisson regression in habitat modelling

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16.1 Mahalanobis distance

Several methods of clustering or classification of multivariate data exist that can be used in habitat modelling. Common for these methods is that they need some sort of measure of distance in the multidimensional space of landscape characteristics in order to assign the most likely class to each unit in the multivariate data set (Lillesand & Kiefer 1994, Kaufman & Rousseeuw 2005). These distances represent the similarity of a specific location or unit to the centre (e.g. mean) of the class it is assigned (Lillesand & Kiefer 1994), where the centre is calculated based on some training data. The training data in habitat modelling is typically geographic locations with known presence of species or ecosystems, or of particular conservation interests. Mahalanobis distance is one of these methods, and its strength is that it accounts for the covariance between the values extracted from the training data when calculating the distance (Manly et al. 2002). Furthermore, the calculation of values is relatively simple compared to other methods of habitat modelling, and it gives easily interpretable outputs. The method has recently been developed to also investigate the importance of different habitat characteristic variables (Browning et al. 2005, Calenge 2006).

The common output from a classification process is a classified map. However, in habitat modelling, we are really not interested in the classified map, but the distances to the centre values from the training data. These distance values will tell us how similar landscape variables such as elevation, forest proportion, and distance to features are to the training data. In the case of habitat modelling, the training data only consist of one class; the presence of the species, biotope, or other feature which we want to model the suitability for over a larger area than the known locations. By using only one class in the training data, all values in the classified output map will have the same class, but the distance map will give information on how similar a specific location is to the training data with respect to landscape characteristics. The larger the distance, the more dissimilar is the landscape at a specific point compared with the training data.

Input variables must be numeric, that is, classified data (e.g. vegetation classes) can not be used. However, it is often possible to convert classified data to numeric by calculating, for example, distances to features or proportions of features inside units (see **Figure 16.1**), or use indices to describe the complexity of habitat classes (Johnston 1998). Furthermore, it is most convenient to use input variables represented as raster data. The training data can be either point locations (e.g. GPS locations, field observations, see **Figure 16.1**) or polygons (e.g. biotope, home range) or a combination of these. It may even be lines. The only thing important is that the training data allows for extraction of values from the input variables, and, if training data come from several different sources, that caution is taken regarding the amount of extracted data from each source. Polygons will give large amount of training data, but are often less accurate than points, which give smaller, but more precise, training data. The training data thus represent the values of the input variables for each observation. These data should have a distribution that does not deviate too much from a normal distribution, and without much correlation. More important is that the distribution is not bimodal. Standardisation of the landscape characteristic variables can be convenient to ensure similar weight of the variables (Calenge 2006).

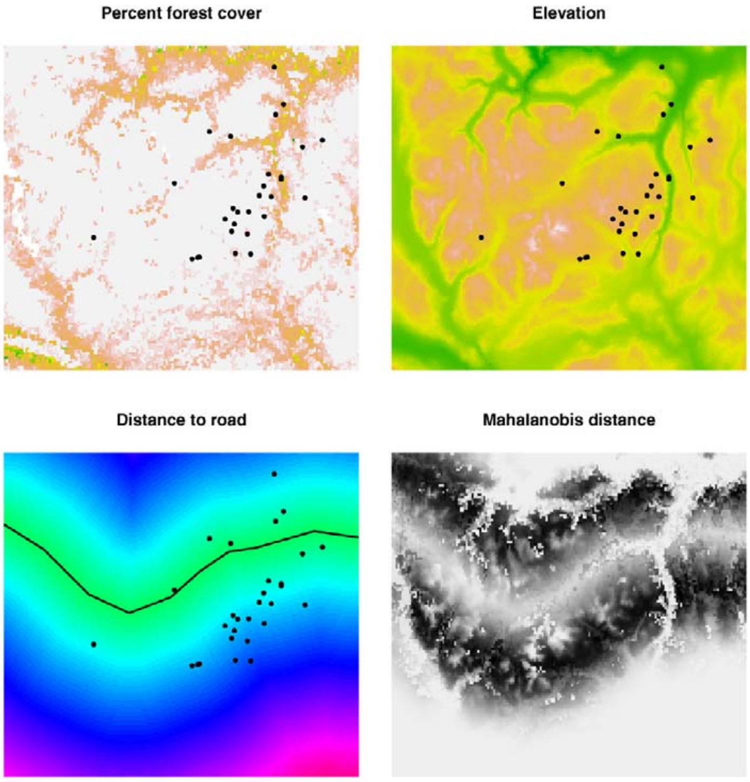

***Figure 16.1** An example of three input layers (percent forest cover, elevation and distance to a road), training points (black dots, that may, for example, represent nest locations of a bird species) and the Mahalanobis distance based on these data. Notice that the two first layers are probably highly correlated. A low Mahalanobis distance (high suitability, black colour) is found at low forest cover, high elevations and at intermediate distances to the road.*

From the training data, the mean values and covariance matrix for the input landscape variables are calculated. Mahalanobis distance (*D^2*) is then calculated as:

$$D^2 = (x - \hat{\mu})'\Sigma^{-1}(x - \hat{\mu})$$

where *x* is a vector of habitat characteristics associated with each pixel, $\hat{\mu}$ is a mean vector of habitat characteristics estimated from the training data, and Σ is the estimated covariance matrix from the training data (Clark et al. 1993, Lande et al. 2003). *D^2* will have high values if the habitat characteristics in the pixel are dissimilar from the training data, and low if it is similar. The distribution of *D^2* is often quite skewed, and a transformation (such as the logarithm of *D^2*) is appropriate to both enhance the visualisation, and to reduce the influence of extremely high values. The output Mahalanobis distance map (**Figure 16.1**) is often regarded as a suitability map for the species, biotope, or other observation unit used as training data (Clark et al. 1993, Lande et al. 2003, Browning et al. 2005, Calenge 2006).

It is possible to combine several species or landscape features in the training data, and try to create, for example, a suitability map for a multi-species approach. However, bear in mind that species differ in their demands for resources, following the principle of the ecological niche (Hutchinson 1957), and the optimum values of landscape characteristics for a combination of species may be far from the optimum of each single species. The same may be true for seasonal differences in habitat use, particularly for migrating species, and for different demographic groups. Thus, it can be more appropriate to model one suitability map for each species, demographic group or season, scale them to the same scale, and overlay the maps in order to identify areas which are important for several different cases.

16.2 Poisson regression

Identifying important areas for species can be solved with different methods of habitat modelling procedures, where the likelihood for a species to use an area is explained by the characteristics of the landscape. Such methods are mostly based on some sort of individual observations of a species, either from relocations of marked individuals, tracks, sightings, location of plant individuals, or locations of nests or dens (Manly et al. 2002). For most of such data, the information is only about where species have been or are present, but not where they are or have been absent (Manly et al. 2002). Therefore, one needs to define some sort of available area. This is often the area where the search for the species have been carried out, or the home range of marked individuals.

Given that the landscape is grouped into areas of different classes (e.g. vegetation classes, elevation ranges), the probability of a species occurring in the classes can be considered a multinomial distribution with one probability for occurrence assigned to each class. Such a distribution, with count data inside each class, can easily be transformed into a Poisson model (Baker 1994, Chen & Kuo 2001, Manly et al. 2002). Briefly, one counts the number of observation inside each landscape class, and calculates the area of each class. The probability of a species occurring in an area is, of course, dependent on the size of the area. Each class is then added as dummy variables in the table. Thus, the table will have one row for each class combination (e.g. three vegetation classes and four elevation classes gives twelve combinations), and columns for number of observation in each class combination, the area of the class, and the classes with 1 and 0 if the row represent the specific class or not. This table can then be analysed by generalised linear models with Poisson family and log link (McCullagh & Nelder 1989). The number of observations is used as dependent variable, and classes are added as explanatory. The area of the classes are best treated as offset variables (Venables & Ripley 2002), but can instead also be added as a fixed factor. In either case, it must be log-transformed to match the log-link in the Poisson model. The importance of classes in explaining occurrence of the species can be evaluated with standard model selection procedure, for example by using an information-theoretic approach (Burnham & Anderson 2002).

In spatial data, autocorrelation between observations is often present, either in time, space or a combination. This often leads to over-dispersion and can influence the uncertainty of the parameter estimates from the model (Lennon 1999, Venables & Ripley 2002). In Poisson regression, this can be accounted for by using a quasi-Poisson family, in which the dispersion parameter (the scale parameter; Venables & Ripley 2002) is estimated, and uncertainty in the parameter estimates (standard errors of the estimates) adjusted according to the dispersion parameter (Venables & Ripley 2002). It is in general recommended to evaluate the dispersion parameter and take the appropriate action before drawing conclusions based on a generalised linear model.

When sampling data, particularly from marked animals, it is common to have data from more than one individual and for a period that include different seasons. With sufficient data, it is possible to evaluate the difference in habitat selection between subgroups of species such as sex or age classes, for different seasons (Henriksen et al. 2003), and even between species. This is simply done by adding a variable such as sex or age as a covariate in the model, and including the interaction between the covariate and the landscape characteristic classes (Henriksen et al. 2003).

Once important variables and covariates are identified and their effects on habitat use are estimated, the probability of occurrence of the species for given landscape characteristics can be predicted for a study area. This can then be used to evaluate the importance of areas for the species. It is important to remember that this only is valid for the demographic group of animals and the seasons from which the data are sampled.

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17 Ecological Niche Factor Analysis

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17.1 Introduction

Many studies of suitable habitat distribution involve large areas for which it may be difficult to gain an overview. With development of GIS (Geographic Information System) many opportunities have become available. Biomapper® (Hirzel et al. 2002) is one method that has been developed to analyze the distribution of suitable areas for species. It has been used for many different organisms such as birds, marine mammals and beetles (Compton 2004, Gallego 2004, Leverette 2004). The overriding principles of the ENFA (Ecological Niche Factor Analysis) are that all variables are ordered as a few uncorrelated factors. One advantage is that only presence data of the species are considered, where the multidimensional space of ecogeographical variables where species are present can be compared to the multidimensional space of available (global) variables.

17.2 Methodology

The method is built on Hutchinson's concept of ecological niche (Hutchinson 1957) and the central concept in the method is marginality and specialization. By definition, marginality is the absolute difference between the global mean and the species mean, and specialization is the ratio between the global variance and the species variance. With the scores from the ENFA it is possible to create a habitat suitability map. Every cell in the habitat suitability map is given a value of habitat suitability index from zero to 100, where zero is bad habitat and 100 very good habitats. The habitat suitability map can be fitted to the purpose of the study, for example through re-classification or filtering. In order to obtain input for ENFA and HS map computation, two types of maps are needed: ecogeographical variable maps and a species map. The ecogeographical maps can be derived from satellite images or from other sources about features in the environment, and the species map is a binary map that show distribution of known location of the species. It should be noted that the species map does not necessarily need to be made for a single species. It could be made as an index or some threshold for a number of species. The ecogeographical maps can, of course, include anything that might be of interest for the species. As long as the maps are in raster format and can be overlaid, they can be applied in the analysis. One advantage is that there is no need for absence data, which may be difficult to verify (MacKenzie 2005). Another advantage is that there is no loss of information due to correlated variables. In methods like the Generalized Linear Model (GLM), the Generalized Additive Model (GAM) and logistic regression a correlated variable that is not significant is removed and does not impact the model. In ENFA, this is not a problem because all variables are ordered in uncorrelated factors according to how much information they explain. If two variables are correlated they both appear in the model with a similar coefficient. Validation of the model is made by Jack-knife cross-validation (Sokal & Rohlf 1995, pp. 820-825).

17.3 Model application: the Hazel Grouse

Application of the method with hazel grouse (*Bonasa bonasia*) gave a marginality of 0.852. Marginality is a value between zero and one and a value close to one indicate a species that utilize a niche that deviates from what is in average available. The resulting specialisation was 2.637 and because this is a ratio between variances, a value >1 indicates a species that utilizes a narrower niche than what is available. This is, however, somewhat more difficult to interpret due to the value between one and infinity, but the inverse of specialisation (tolerance) is a value between zero and one, with specialists indicated by values close to zero. From the results of the ENFA and the following habitat suitability map it is straightforward to get the patch structure. The patch structure may be classified in poor habitat, suitable habitat and core habitat, and from this, it is easy to draw geometric measures such as area and perimeter. With these measures further calculation of perimeter-area ratio or fractal dimension give valuable measurements of shape complexity.

17.4 Conclusion

This method gives many of the fundamental variables that are needed in other models, such as metapopulation models. It is also possible to analyze niche differentiation, either between competing species or between predator-prey species. When it comes to management or conservation strategies the method is very suitable in an initial state to find areas of interest and to find amount of suitable areas for a species.

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18 How to allow for dependent observations by the use of Mixed Models

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18.1 Autocorrelations and dependencies within statistical units

One of the main assumptions for using classical analysis methods such as Generalized Linear Models (GLM; linear regression, ANOVA, logistic regression, etc.), is the independence of observations (e.g. Crawley 2005). Such data are easily produced in properly designed laboratory studies or other kinds of controlled experiments. In observational studies, however, investigators have to record naturally occurring events, for instance data points sampled through time or through a geographic range. By their nature, spatial and temporal data are almost always autocorrelated; two observations close in time or space are likely to be more similar than another random pair of observations. Moreover, if two data points are sampled from the same individual, they are more likely to be similar than data points sampled from two different individuals.

Recent developments in radio-telemetry and Global Positioning System (GPS) technology has yielded large amounts of data sampled through a temporal and spatial range and with multiple relocations taken from each individual animal in the study. Despite violating the assumption of independence, GLM is still a common way to analyze such data, often by means of logistic regression within the framework of Resource Selection Functions (RSFs; Manly et al. 2002). In a logistic regression model, the probability of presence is estimated, on the basis of observed presence/absence data. Most telemetry/GPS-data, however, contain only presence, and not absence, of the focal species. To bypass this problem, available points, or so-called “pseudo-absences”, are then randomly selected from the geographical range available to the animals (Manly et al. 2002). The number of available observations chosen is somewhat arbitrary in each study, but will always increase sample size. If this is not accounted for, we will end up doing our analysis based on pseudo-replicates (Hurlbert 1984). Generally, the effect of ignoring autocorrelations and dependencies is to increase type I error rates (i.e. standard errors of coefficients in models are biased low), especially if the number of relocations varies greatly among animals (Erickson et al. 2001).

In a recent paper, Marzluff et al. (2005) introduced the Resource Utilization Function (RUF) as a new technique of analyzing spatial data. A RUF expresses the correlation between an animal’s utilisation distribution (e.g. a Kernel; Kernohan et al. 2001) and sets of spatially defined resources on a cell-by-cell basis. Obviously, such an approach may also imply an artificially large sample size, where the number of data points is merely decided by an arbitrary choice of grid cell size made by the researcher. Correctly, Marzluff et al. (2005) used a multiple regression technique adjusted for spatial dependencies to analyze their data.

18.2 Mixed Models

In a mixed model, each factor can be defined as either fixed or random (Pinheiro & Bates 2000). A fixed factor is a factor where the levels of the factor are all levels that are of interest to the researcher. A random factor is any factor that is measured in the study by items, levels, subjects, etc. which (conceptually) is randomly selected from a larger set of possible items, levels, etc. any of which could have been used in the study. With a random factor we can thus

account for dependencies that exist within individuals (or study areas, years, etc.) by considering each animal as the experimental unit in the analysis (Otis & White 1999).
If the data also contain spatial and/or temporal autocorrelation, it should be dealt with by assigning it a proper covariance structure (Littell et al. 1996). Different types of software have gradually implemented mixed models as part of their repertoire. In the SAS macro GLIMMIX (Littell et al. 1996) it is possible to make linear models including both fixed and random factors for both temporally (i.e. linear or one-dimensional) and spatially (i.e. two-dimensional) autocorrelated data. Using GLIMMIX, one can assume a long list of error distributions, including normal, poisson, and binomial distributed responses. The corresponding link functions are then identity, log, and logit, respectively.

Below, I show an example of a GLIMMIX script where “presence” is a binomial response variable, “temperature” and “season” are fixed effects, and “individual” is treated as a random effect. The spatial structure, represented by geographical coordinates (i.e. pairs of x’s and y’s) is here defined as “spherical” (Littell et al. 1996). An R or S-plus equivalent to the GLIMMIX-macro would be the glmmPQL, glmmML, or lmer functions in the three libraries MASS, glmmML, and Matrix, respectively (Pinheiro & Bates 2000).

```
%glimmix (data=workspace. datafile, procopt=method=reml, stmts=%str
 (class individual season;
 model presence = temperature season;
 random individual / type = sp(sph)(x y);
),
 error = binomial,
 link = logit);
run;
```

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19 Possibilities and limitations of using habitat modelling in the real world – results of group discussions

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19.1 Introduction

The presentations given at the workshop laid the groundwork for the group discussions that followed. Groups were comprised of representatives from management and research, to ensure that both standpoints were addressed on each topic.

Two main topics were discussed by the groups: (1) Application of habitat modelling in the real world, and (2) Model approaches and data needs. Groups were given a set of specific themes and questions within each broader topic. Each group summarized their conclusions in front of the entire group, and discussion followed. The following summarizes the main points brought out in this exercise.

19.2 Topic 1: Application of habitat modelling in the real world

19.2.1 Goals and needs

Habitat modelling is a useful tool for measuring progress in reaching national and/or international goals for conserving biodiversity and sustainable development. Since empirical biological data are generally rare, and expensive to obtain, modelling is the only practical method for assessing key parameters, particularly at larger scales. Models can be used as tools for predicting the consequences of given alternatives for human activities, such as new infrastructure or housing areas, and allow planners to find solutions that have minimal impact on the environment. In most, but not in all cases, however, models must be verified through research, and relevant data must be collected before and after the impact in order to test the assumptions of models.

In the group discussions, there was a general consensus that there is a large need for incorporating habitat models in physical planning, but there is much uncharted territory, in both the figurative and literal senses. First of all, it is often difficult for those responsible for physical planning and landscape management to express what needs they have, when they are uncertain as to what is available or even desirable. As a result, planners may not even think to utilize such models in their projects and activities. In many cases, agencies tend to plan and carry out projects on the basis of earlier experience, without giving adequate consideration to consequences under the assumption that results will be satisfactory. Ideally, models should be adaptable and dynamic, such that different scenarios and outcomes can be predicted by adjusting relevant variables to reflect changes over time and space, and model the effects of different measures and activities.

There is a clear need for interaction between planners (users) and modellers to determine the appropriate resolution and quality of data needed for typical applications. Scale is an issue here – do we need data at a higher resolution than presently available through existing data bases (e.g. CORINE)? In many cases, we lack the data needed to address specific concerns. In addition, when data is available it may not correspond to the scale for which it is needed. It is therefore useful to identify the needs for data and the appropriate scale(s) for which they are to be used. Furthermore, planners may need assistance in understanding limitations and possibilities of models and help in formulating appropriate and concise questions. All this requires improved communication and understanding between modellers and model users.

Models can be used for predicting the impacts of human activities, and are essential for good planning and management. They can also be used to identify areas of special environmental concern, such as critical habitats for red-listed species. In addition, researchers can use models as a tool for effective data collection. Ideally, models can be improved as more data become available, and thus there is a need for data bases that can be used for this purpose. Implicit in this is a need for standardization and consolidation of programming tools for wide application; for example, meteorologists use prognosis tools that are both systematic and internationally compatible. At the moment, we have no unified approach for modelling terrestrial ecosystems and their components.

There are generally quite clear goals and needs for predictive applications to ensure the viability of red-listed species over time, to protect nature reserves, or to determine the effects of infrastructure projects. Managers and planers need to know, if possible, what the threshold values are - given a certain scenario, what are the chances that a given species will still be present in ten years time?

In Sweden, the so-called “Base inventory” (“Basinventering”) has been ordered by the Nature Protection Agency in order to obtain a biological inventory of national parks, nature reserves and other conservation areas. The next stage in this work could address national interests in relation to these. A level above this could be more specific, and address community planning relative to requirements for environmental assessment in order to better focus on actual data and modelling needs. Such a process should be transparent, such that the parameters selected are known.

Biologists responsible for research and monitoring of species and habitats also require data for their work. Politicians demand to know what consequences our activities have on the environment, and require that environmental and societal interests be balanced through predictive models. Planners require the best available input from scientist in order to properly and responsibly address and meet the demands of society and the environment in their work.

Modelling can be used to determine which data are needed, and at what scale, for particular uses. It can thus be an interactive process, which can also lead to new classification schemes for habitat mapping. In this regard we can focus, for example, on special and prioritized types of natural habitats in this context. We could model particular species and/or hot spots through the use of empirical data, which are in turn used to construct classes to be used in models. In such cases, it may be advantageous to collected data in as raw a format as possible, i.e. data on the distribution of particular species or species groups which can then be classified according to their associations with map variables. Local data can be used to make broader scale classifications, when and where possible, which can then be used in national classification schemes. In other words, there is a need to use point-sampling data to make broader generalizations at larger levels of scale, relative to the needs associated with broader scale planning and construction activities.

We can use modelling to predict effects of climate change, construct time series (such as what happens if tree lines move, what happens when red-listed southern species move north?). For planning purposes, it might be better to model key aspects of important biotopes, such as dead wood, which may include as much as 25% of rare forest species. This can be an effective tool in defining old forest areas. By plotting empirical data, we can look at clusters and create habitat models for identifying potential habitat features which have not yet been surveyed. These areas can then be surveyed for species presence, and can also be considered as potential habitat even though red-listed species may not be present.

19.2.2 Application in planning

In general, models must be adaptable to user needs. These must be able to identify key habitats for species of concern, which can in turn be mapped and used for planning activities. Models should also be capable of providing reliable and comprehensible predictions of impacts of proposed activities. Results should be robust but not trivial. Input data in empirical models should also provide important time series data for monitoring purposes. In road planning, for example, important parameters must be identified in order to define potential consequences of different possible corridors.

In many cases, planners must subjectively balance different national needs without the aid of even simple models. In this regard, expert models can provide associations between species and habitats which can be quite useful. Expert models can be used when empirical data is not available as a starting point. These can be replaced or adjusted as empirical data become available. Maps for specific species can be made as layers in GIS, and thus identify critical areas where many species overlap.

Habitat Suitability Index models (HIS) have the advantage that these assign values for map units, but are rather subjective and based upon uncertain assumptions. Such models can be used if they have been tested and approved. However, assumptions must be stated clearly, and test criteria and associated data needs must be clearly defined.

Scale is a challenge when creating models. Microhabitat features that are critical for species of high concern may not be included in large scale models. In addition, large scale models may work for one region, but may not be transferable to other regions.

19.2.3 Communication

There is clearly a need for improved dialogue between modellers and users. At the moment, we lack a vision on how to best cooperate. In Scandinavia, we have no clear overview over those involved in this field, although there are relatively few that work with modelling in Norway and Sweden. Time is a factor, as users and researchers are very busy people, and there is little time for networking and building cooperative relationships. There is clearly a need for increased competency, but an even greater need for data.

Researchers need to communicate how their data and models can assist planners and managers relative to their responsibilities. At the same time, users must specify what their needs are, and how they hope to use models and biological data in their work. An interdisciplinary approach, where users and modellers interact, is the best possible scenario for the development of useful and meaningful models that simultaneously satisfy scientific and user interests.

Sector authorities and biologists must come together and identify needs. Biologists must learn about management challenges and planning processes in order to serve these interests. Users must understand the possibilities and limitations of models, and be willing to work with biologists in their creation and use. We need specialists that can bridge the gap, and that can adapt models according to the applications they are intended for.

In the world of science, applied science is often regarded as of less merit than theoretical science. In this regard, modelling can also be viewed as an attractive field for researchers that wish to receive academic merit while working on real-world applications. Since models require biological data for verification, this provides an impetus for systematic data collection which can then be used in long-term monitoring schemes, which in turn can provide information on relationships between species and habitats at different levels of scale. These data can in turn be used for adapting models and for making more precise predictions.

Why haven't models been used more in planning? Is it due to a lack of competence or understanding among decision makers and planners, or is it because modellers do not understand the need of the former? GIS experts will have quite clear parameters which can be measured and quantified. Biologists, on the other hand, often tend to qualify their evaluations and avoid making generalizations due to the inherit complexity of organisms and ecosystems. There is, therefore, a need for the differing *modus operandi* between these groups of professionals. It is essential that biologists actively participate in planning processes, and that there is strong interaction between users and biologists in this context.

19.2.4 Data availability

There are many databases on species, habitats and human activities that affect these, but databases vary with respect to function, platform, software, format and classification schemes that are used and often do not support data merging from different sources. We do not have a complete overview of available databases in Scandinavia at present. A first step should be the creation of national meta-databases that provide complete and updated overviews over existing databases on relevant topics (see topic 2). In the future, we should strive for compatibility of databases and classification systems across international borders (at least in the Nordic region).

Research requires access to data for testing models, and has therefore a need for both observational data as well as spatial data. Species Data Banks in Sweden and Norway are addressing this issue. There is a need for close cooperation between these institutions and academic institutions, in order to identify and develop applicable models. A system, which allows for direct access and input of map-referenced observational data coupled to map coverage data would be ideal. Such a system would allow researchers to test models with empirical data. Such a system will require coordination, quality control, and delegation of responsibility for subsets of data among researchers.

At present, another system, known as HEUREKA, is available for Swedish landowners as a web application for simulating the outcomes of different forest harvest regimes on the bases of forest stand maps. However, the underlying data are not available to users in this system.

Knowledge databases, for example a "Species Fact Database", that contain comprehensive information regarding species and their habitat associations by region should be prioritized. Gaps in knowledge should also be identified. Such an undertaking is already underway in Fennoscandia – a knowledge database on species associated with dead wood.

19.3 Topic 2: Model approaches and data needs

19.3.1 Biological data (model parameters)

There are many sources of biological data in Scandinavia. There are diverse monitoring programs that are funded by management agencies that collect data according to a variety of sampling schemes of varying depth and breadth of detail. Researchers and non-governmental organizations have gathered a wealth of data on distribution, abundance and habitat relationships of numerous species through a variety of methods. Data gathered by researchers or private individuals are often not available for use outside of the projects for which they were gathered. In addition to modern sources of data, historical data on abundance and occurrence are hidden in older publications, museums, and the like, and much remains to be uncovered.

Different kinds of data have been and are being gathered in our region. Degree of detail, parameters included, vary greatly from database to database, since data demands vary between projects and programs. Presence/absence data may or may not include measures of sampling effort. Methodology may be systematic or random, and coverage varies from small localities to larger regional scales. Correlative models depend upon reliable map-referenced data on the distribution of species. Good examples include the Bird Atlas, Species Data Bank, etc. There are a multitude of databases, including those with verified observations, but quality varies.

There is a need for broader coordination of monitoring and research on species of special concern. At the same time, we must identify where the gaps exist in our knowledge. A strategy is therefore needed regarding which species and issues to prioritize, and how.

Meta-databases will help to gain an overview of existing databases and can help in making these data available to researchers and the authorities. In this context, each database should contain metadata which include information on sampling effort, sex, age, reproduction and other variables. A Scandinavian clearinghouse or meta-database similar to and compatible with the Global Biodiversity Information Facility (GBIF: http://www.gbif.org/) should be considered. The GBIF is an independent international organisation whose overall mission is to work with its partners (countries, international organisations, natural history museums, herbaria, the scientific and IT communities, and the international biodiversity-related conventions) to provide free and universal access to the world's primary biodiversity data.

At present, we do not have such a portal/meta-database/clearinghouse in either Norway or Sweden. We should not only have an overview regarding existing databases, but also ongoing projects. Since work needs to be done in both countries, this should be coordinated where possible. Species Data Banks in both countries could play an important role here, in close co-operation with the management and map services in both countries. A lot of data is collected in various projects, and much of it will languish if it is not made available for wider use.

A Scandinavian portal, in Nordic languages, would be useful in our region, but would be of limited value internationally. Thus, we should strive to make important databases available in the international language of English in order to address global issues. Cooperation at this level may also facilitate common solutions and avoid the pitfalls of "re-inventing the wheel" where proven systems and methods exist. At the same time, our efforts can be added to a global understanding of ecosystem relationships and changes, as well as methodology.

19.3.2 Uncertainty and error

Expert models are based upon the assumptions of experts that a species or group of species are present in a given habitat. The habitat itself may also be described in a subjective manner. Such models can be quite useful, but there is a need for testing and verification with empirical data. Map information must be interpreted relative to known habitat relationships – can we identify or correlate important microhabitat features in remote sensing data? As such, expert models can be difficult to ground-truth and validate. However, it is essential that a system be devised which can filter out gross errors

Empirical data are inherently superior in many cases to expert models, and can be used to validate the latter. This approach also has its drawbacks, however. Field collection of data is often an expensive undertaking, in terms of human effort and monetary investment. Design of studies and programs determine the kind of data that will be collected. Binary data (presence/absence) is inferior to data that can provide variances in predictions, since presence alone is not necessarily a measure of habitat quality. In addition, there is inherent uncertainty relative to the classification of pixels into discrete habitat units.

Managers and researchers operate under different conditions and have different requirements for data quality. Managers often need quick and clear answers, which researchers may not be able to readily provide. There will always be uncertainty in any model which attempts to predict the future on the basis of selected input variables. Models can give us clues as to probable outcomes, based on best available knowledge. Experience with other forms of prognosis modelling from meteorology and economics, for example, should teach us that our models may also fail to predict real consequences, and we must resign ourselves to this. Managers and planners must apply a measure of common sense and interpret models accordingly. Uncertainty will generally increase with the complexity of data and the detail of knowledge needed in a specific situation. The variance in predictions should therefore be indicated, where possible. Alternative scenarios should be presented with best and worst case scenarios. Models must be transparent relative to their limitations and assumptions, and decision-makers must understand these when interpreting predictions they make.

19.3.3 Generalization – extrapolation

It is important to remember here that models are simplifications of reality. In addition, models are often designed for or adapted to specific species or species groups in selected localities. As a result, it can be difficult to make generalizations regarding species communities or the entire landscape on the basis of these. Generalists will dominate most survey data, since they are found in many kinds of habitats. Generalists may add to overall biodiversity, but species with highly specific needs will be more important from a qualitative viewpoint. Umbrella species can be used to assign values to specific habitats/map values. Umbrella, focal or flagship species can be valuable for communicating conservation values with which they are associated. In some cases, it may be useful to construct “phantom species” which incorporate a superset of variables important for plant-animal associations of particular conservation value. In this regard, modellers should be careful in how organisms are classified. It is more instructive to classify associations rather than groups based on taxonomy. In addition, we can classify organisms relative to their response to human activities, e.g. species that are vulnerable to a particular activity such as road construction.

Scale properties (extent and resolution in time and space) are important issues here, as is the level of detail in data. There is a growing need for modelling landscape functionality and effects, including the effects of fragmentation of important habitats on area-sensitive species. We need to identify criteria for defining landscape types that are relevant for such issues. Classifications should be relevant for approaches which seek to address landscape-level processes, and comparable at broader levels of scale.

19.3.4 Validation

It is often difficult or impossible to validate models that are created only for planning purposes or that use ecological profiles, “phantom species” or ecosystem function. Validation, however, may be important to assess the accuracy of model predictions for use in management and planning. Validated models can effectively increase the precision of predictions and decrease planning and management expenditures. The broader applicability of a given model can be tested by assessing its ability to correctly predict relationships outside the study area in which it was derived. For empirical models, a data set can be divided (by area or time period) and groups tested against each other to assess their ability to predict habitat use for a given species. Expert models can also be tested by using empirical data in order to improve rules regarding habitat parameters associated with particular species. Correlative models are difficult to test, however, since increased sample size only gives more observation points. In this case, expert knowledge can be used to assess the validity of such models.

An important aspect of this issue is the need for quality control of input data and model rules. In general, the model quality will be enhanced by large sample sizes and correspondingly smaller confidence intervals. Quantitative models can be controlled through the use of algorithms and statistics in order to flag obvious errors relative to known habitat requirements for species. Variation in observer quality can be analysed by assessing reporting patterns. Long-term monitoring programs can provide larger data sets over time that can allow for adaptive refinement and adjustment of models as data sets grow, as is the case with the Swedish National Inventory of Landscapes (NILS) database. Where applicable, research data on animal movement patterns (i.e. radio telemetry) can be used to assess model predictions. Qualitative models can be subjected to the scrutiny of experts to ensure their validity.

20 Conclusions

There are clearly many opportunities and challenges regarding the application of habitat modelling here in Scandinavia. Participants agreed that this tool has wide application, from assisting researchers in measuring, identifying and monitoring biodiversity to predicting the impacts of any given infrastructural project on fauna and flora. As time progresses, tools and techniques will continue to be improved and refined as more data become available.

We see that spatial modelling can be an important predictive tool for planners, and is also useful for mapping and researching biodiversity. There is, however, a need for better communication between scientists and planners in order to optimize our capabilities. Researchers often develop models and techniques that are very specific for their projects, and may or may not have broader application for planners. Planners, on the other hand, may not be aware of existing tools that can aid them in their activities. Clearly, we must focus on better communication and awareness between these groups. Ideally, we need an intermediary group of specialists that posses the technical and scientific capabilities and knowledge necessary for integrating disciplines that can bridge this gap and facilitate communication and cooperation necessary for development and application of tools and technologies.

It is imperative, however, that we make full use of existing data and technologies through a unified approach. Programming tools, classification systems and methodologies must be standardized and consolidated in a systematic fashion to ensure compatibility across national boundaries. The broad similarities between Norway and Sweden can facilitate such standardization. However, a Scandinavian system must also be based upon international standards, and there is thus a need for consensus at the global scale with regard to tools used in mapping and modelling ecosystem and biodiversity components at different scales. Finding a standard that can adapt to changing technologies and improved knowledge that can be applied globally will be no easy task, and will require a concerted effort in the international arena.

Inherent in such an approach is the need of broad availability of standardized data. National meta-bases should be created and made available for all users. Ideally, a single map system that integrates biological data in Sweden and Norway would be the most efficient approach. Such a system could incorporate and be integrated with internationally recognized systems and should therefore be available in English, the international language of science and technology. We suggest the establishment of a Scandinavian clearinghouse or meta-database that would be compatible with the GBIF. Such a system should be open-ended and interactive and allow for new data and technologies to be adapted and incorporated as these become available.

Models are only as good as the data and assumptions they are based upon. Empirical data vary in complexity from binary to complex layered and multivariate arrays. Empirical data can be used to build and validate classification schemes based on remote sensing data. Such data can also be used to test the assumptions and accuracy of expert models. By consolidating related data across projects and administrative boundaries, models and systems can adapt and evolve over time, and data quality can be enhanced as sample sizes increase and relationships to map data become clearer through the process of validation.

This workshop represents a starting point for the integration of data and methodologies on the Scandinavian Peninsula and increased cooperation between planning and research sectors. This field is relatively complex and technically and methodologically advanced, and there are in reality few practitioners in our two countries. Funding is also limited, and we must make the best possible use of our resources and knowledge to the benefit of biodiversity conservation and landscape management and planning. By consolidating our skills and knowledge through regular communication and cooperation, not only within Scandinavia, but also at the European and international level, our efforts can be enhanced. In a changing world that is increasingly impacted by human activities, a focused and unified effort may indeed be crucial for the long-term health of the global environment and, ultimately, the human race.

Appendix 1 – Workshop program and list of participants

Program

(official languages were Swedish and Norwegian)

Tisdag 14. februari

19:00 Ankomst til Sunnersta (http://sunnersta.mygiicms.com/)
20:00 Middag
21:00 Sosial kveld. Bli kjent.
21:30 Ankomst norske deltakere

Onsdag 15. februari

07:00	Frokost	
08:00	Velkommen, bakgrunn, forventninger, gjennomføring, presentasjon	Scott Brainerd, Andreas Seiler, Leif Kastdalen, deltakerne
08:30	***Seksjon 1: Habitatmodeller - muligheter og begrensninger***	
09:00	Romlige modeller – en oversikt	Michael Gontier
09:35	Pause	
10:00	***Seksjon 2: Brukerperspektivet - behov og utfordringer***	
10:00	Habitatmodeller i MKB/SMB	Berit Balfors
10:35	Transportsektoren	Anders SjölundandBjørn Iuell
10:50	Naturforvaltningen	Person fra DNandEbbe Adolfson
11:35	Informasjon/formidling – Artsdatabanken	Jon Atle KålåsandOskar Kindvall
12:00	Lunsj	
13:00	***Seksjon 3: Kartdata. Eksisterende databaser, fremtidige muligheter***	
13:00	Modellering i arealplanering	Ulla Mörtberg
13:30	GIS data i Sverige og Norge	Kerstin Nordström /Conny Jacobsen, Leif Kastdalen
13:55	Pause – kaffe	
14:10	Datafangst med fly- og satellittsensorer	Leif Kastdalen
14:45	**Gruppdiskussioner 1** Fokus på användning: målsättningar, behovandkrav, kommunikation	Arbeidsgrupper
17:45	Frukt - kaffe	
18:00	Presentasjon, sammenfatting	
19:00	Middag	
20:30	Sosialt / software fordypning	

Torsdag 16. februari

07:00	Frokost	
08:00	***Seksjon 4: Habitatmodeller – eksperttilnærmin*g**	
08:10	Databaser – systematisering av kunnskap	Kjetil Bevanger
08:35	Regionale modeller – viltkartlegging	Leif Kastdalen
09:25	Pause	
09:35	Modelltilnærming för vägplanering	Görgen Göransson
10:10	Habitatmodeller og planlegging	Per Angelstam
10:55	Pause	
11:15	***Seksjon 5: Habitatmodeller - bruk av empirisk data***	
11:15	Stedfestede punktobservasjoner – ressursseleksjonsfunksjoner (RSF)	Unni Støbet Lande/Andreas Seiler
11:35	Fra punkt til areal– Poisson og Mahalanobis tilnærminger	Ivar Herfindal
12:20	Lunsj	
13:20	Økologiske nisje faktor analyse med Biomapper	Jonas Sahlsten
14:00	Mixed models - hvordan ta hensyn til ikke-uavhengige data i modellbyggingen?	Hege Gundersen
14:30	**Gruppdiskussioner 2** Fokus på metoder og modelltyper	Arbeidsgrupper
17:30	Kaffe, frukt	
18:00	Presentasjon, sammenfatting	
19:00	Middag	
20:30	Sosialt / software fordypning	

Fredag 17. februari

07:00	Frokost	
09:00	Sammenfatting and öppen diskussion	alla
10:30	Pause	
11. 00	Viktigaste Budskap och lärdomar	alla
11:30	Outlook into the future	Scott, Leif, Andreas
11:45	Lunch	
	Hemresa	

List of participants

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Appendix 2 – Nomenclature of the Swedish Land Cover data (SMD)

Reference: Ahlcrona, E. 2003: Nomenclature and class definitions. Appendix 1, Swedish Corine Land Cover Data, Product Specification, Lantmäteriet.

1. ARTIFICIAL SURFACES
1. 1 Urban fabric
1. 1. 1 Continuous urban fabric
1. 1. 2 Discontinuous urban fabric
1. 1. 2. 1 Discontinuous urban fabric with more than 200 inhabitants
1. 1. 2. 1. 1 Discontinuous urban fabric with more than 200 inhabitants with minor areas of gardens and greenery
1. 1. 2. 1. 2 Discontinuous urban fabric with more than 200 inhabitants with major areas of gardens and greenery
1. 1. 2. 2 Discontinuous urban fabric with less than 200 inhabitants
1. 1. 2. 3 Rural fabric with open plots

1. 2 Industrial, commercial and transport units
1. 2. 1 Industrial or commercial units
1. 2. 2 Road and rail networks and associated land
1. 2. 3 Port areas
1. 2. 4 Airports

1. 3 Mine, dump and construction sites
1. 3. 1 Mineral extraction sites
1. 3. 1. 1 Sand and gravel pits
1. 3. 1. 2 Other mineral extraction sites
1. 3. 2 Dump sites
1. 3. 3 Construction sites

1. 4 Artificial non-agricultural vegetated areas
1. 4. 1 Green urban areas
1. 4. 2 Sport and leisure facilities
1. 4. 2. 1 Sport grounds, shooting ranges, motor, horse and dog racing tracks
1. 4. 2. 2 Airfields (grass)
1. 4. 2. 3 Ski slopes
1. 4. 2. 4 Golf courses
1. 4. 2. 5 Non-urban parks
1. 4. 2. 6 Camping sites and leisure home development

2. AGRICULTURAL AREAS
2. 1 Arable land
2. 1. 1 Non-irrigated arable land
2. 2 Permanent crops
2. 2. 2 Fruit trees and berry plantations
2. 3 Pastures
2. 3. 1 Pastures

3. FORESTS AND SEMI-NATURAL AREAS
3. 1 Forests
3. 1. 1 Broad-leaved forest
3. 1. 1. 1 Broad-leaved forest not on mires or bare rock
3. 1. 1. 2 Broad-leaved forest on mires
3. 1. 1. 3 Broad-leaved forest on bare rock

3. 1. 2 Coniferous forest
3. 1. 2. 1 Coniferous forest not on mires or bare rock
3. 1. 2. 1. 1 Coniferous forest on lichen-dominated areas
3. 1. 2. 1. 2 Coniferous forest not on lichen-dominated areas
3. 1. 2. 1. 2. 1 Coniferous forest, not on lichen-dominated areas, 5-15 m
3. 1. 2. 1. 2. 2 Coniferous forest, not on lichen-dominated areas, >15 m
3. 1. 2. 2 Coniferous forest on mires
3. 1. 2. 3 Coniferous forest on bare rock

3. 1. 3 Mixed forest
3. 1. 3. 1 Mixed forest not on mires or bare rock
3. 1. 3. 2 Mixed forest on mires
3. 1. 3. 3 Mixed forest on bare rock

3. 2 Shrub and/or herbaceous vegetation association
3. 2. 1 Natural grassland
3. 2. 1. 1 Grass heath
3. 2. 1. 2 Meadow
3. 2. 2 Moors and heath land
3. 2. 4 Transitional woodland/shrub
3. 2. 4. 1 Thickets
3. 2. 4. 2 Clear-felled areas
3. 2. 4. 3 Younger forest

3. 3. Open spaces with little or no vegetation
3. 3. 1 Beaches, dunes, and sand plains
3. 3. 2 Bare rock
3. 3. 3 Sparsely vegetated areas
3. 3. 4 Burnt areas
3. 3. 5 Glaciers and perpetual snow

4. WETLANDS
4. 1 Inland wetlands
4. 1. 1 Inland marshes
4. 1. 2 Peat bogs
4. 1. 2. 1 Wet mires
4. 1. 2. 2 Other mires
4. 1. 2. 3 Peat extraction sites
4. 2 Coastal wetlands
4. 2. 1 Salt marshes

5. WATER BODIES
5. 1 Inland waters
5. 1. 1 Water courses
5. 1. 2 Water bodies
5. 1. 2. 1 Water bodies, open water area
5. 1. 2. 2 Water bodies, vegetation covered water area
5. 2 Marine waters
5. 2. 1 Coastal lagoons
5. 2. 2 Estuaries
5. 2. 3 Sea and ocean
5. 2. 3. 1 Sea and ocean, open water area
5. 2. 3. 2 Sea and ocean, vegetation covered water area

NINA Report 195

ISSN: 1504-3312
ISBN: 978-82-426-1755-2

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Grafic design: NINA
Cover photoes: Per Jordhøy, Børre Dervo,
Knut Kringstad, Tycho Anker-Nilssen NINA