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Biodiversity in environmental assessment—current practice and tools for prediction

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Abstract

Habitat loss and fragmentation are major threats to biodiversity. Environmental impact assessment and strategic environmental assessment are essential instruments used in physical planning to address such problems. Yet there are no well-developed methods for quantifying and predicting impacts of fragmentation on biodiversity. In this study, a literature review was conducted on GIS-based ecological models that have potential as prediction tools for biodiversity assessment. Further, a review of environmental impact statements for road and railway projects from four European countries was performed, to study how impact prediction concerning biodiversity issues was addressed. The results of the study showed the existing gap between research in GIS-based ecological modelling and current practice in biodiversity assessment within environmental assessment.

Keywords: Environmental impact assessment; Biodiversity assessment; GIS-based ecological modelling; Prediction tools; Strategic environmental assessment; Road and railway projects

1. Introduction

Habitat loss and fragmentation are major threats to biodiversity (Fahrig, 1997; Wilcove et al., 1998). Many planning decisions on infrastructure and other developments cause, when implemented, fragmentation of natural habitats. This results in both habitat loss and isolation, and often also causes habitat degradation (Opdam and Wiens, 2002). Infrastructure projects

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contribute significantly to such problems; of particular importance are the effects of habitat isolation due to barrier effects, and habitat degradation, caused by e.g., noise and air pollution (Forman, 2000; Seiler, 2002; Trocmé et al., 2002). Habitat loss affects the long-term viability of populations, which will be lower when populations become too small and are eventually threatened by local or more regional extinction (Opdam et al., 2002). Habitat loss and fragmentation may, in turn, lead to impacts on biodiversity at genetic, species and ecosystem levels, which are levels that should be considered in Environmental Impact Assessment (EIA, Slootweg and Kolhoff, 2003).

EIA and Strategic Environmental Assessment (SEA) are essential instruments to assess impacts from infrastructure and other developments. Further, ecological issues have been an integral part of the EIA process since it was incorporated in The National Environmental Policy Act in 1969, in the USA. In Europe, the text of the EIA directive, published in 1985 (Official Journal of the European Communities, OJ, 1985) and amended in 1997 (OJ, 1997), specifies that impacts on flora and fauna need to be considered. More recently, the directive on the assessment of the effects of certain plans and programmes on the environment, or SEA directive (OJ, 2001) specifies that biodiversity as well as flora and fauna must be part of the assessment. Another important aspect of the consideration of biodiversity in the EIA process is related to nature conservation and its regulation, mainly concerning protected sites and species.

A number of convention bodies as well as international and national organisations and administrations have influenced and facilitated the integration of biodiversity issues in the EIA process, and in some cases published specific guidelines. In the text of the Convention on Biodiversity (CBD), article 14 stipulates that each contracting Party shall introduce EIA procedures for projects that are likely to have impacts on biological diversity (UNCED, 1992; OJ, 1993). The CBD defines biological diversity as “the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems”, including diversity within species, between species and of ecosystems. Other convention bodies like Ramsar (UNESCO, 1971), Bern (Council of Europe, 1979) and Bonn (UNEP, 1979), have also been influential on the topic. Specific guidelines on biodiversity/ecological assessment issues were published by, for instance, the Council on Environmental Quality (CEQ, 1993) in the USA, Department of Transport (1993) in the UK, Canadian Environmental Assessment Agency (CEAA, 1996), Swedish National Road Administration (SNRA) and Swedish National Rail Administration (1996), World Bank (1997, 2000), International Association for Impact Assessment (2001), Direction régionale de l’environnement de Midi-Pyrénées (DIREN, 2002) in France, and SNRA (2002). Many guidelines as well as the CBD (CBD, 2004) recommend adopting and applying an ecosystem approach whenever appropriate.

In Europe, a report on the effectiveness of the EU directive on EIA (European Commission, 2002) concluded that little information is available on how biodiversity issues are addressed in practice. The main qualities of and failures to incorporate biodiversity and ecological issues in the EIA process have, however, been studied in Europe (Treweek et al., 1993; Thompson et al., 1997; Byron et al., 2000) and in the USA (Atkinson et al., 2000). Some of the overall conclusions concerned the vagueness and descriptive nature of assessments, the focus on protected areas and protected species, the confinement to single development actions and on-site changes, and the lack of assessment at the ecosystem level and at the spatial and temporal scales of ecological processes (Treweek et al., 1993; Byron et al., 2000; Atkinson et al., 2000; Geneletti, 2002; Slootweg and Kolhoff, 2003). Further, according to several authors (Treweek et al., 1993; Thompson et al., 1997; Byron et al., 2000; Atkinson et al., 2000; Geneletti, 2002), there is a lack of adequate methodologies for accurate, systematic and quantified predictions of impacts on biodiversity.

However, predictive ecological models have been developed within research disciplines like landscape ecology, spatial ecology and conservation biology (Hanski, 1994; Guisan and Zimmermann, 2000; Akçakaya, 2001; Opdam et al., 2002; Lehmann et al., 2002a; Scott et al., 2002) including applications such as strategic conservation planning and forest management (Margules and Pressey, 2000; Gutzwiller, 2002; Angelstam et al., 2005). By applying such tools, the distribution of valued biodiversity components, for example habitats, species and communities, can be modelled and visualised in GIS, in a format suitable for scenario-testing. Such models have potential for providing quantitative and spatially explicit predictions of impacts on biodiversity components.

The aim of this paper is to identify needs and potential for future methodological improvements in the prediction of impacts on biodiversity within EIA and SEA. The paper presents an overview of GIS-based ecological models, which offer potential application as prediction tools in biodiversity assessment, in particular to assess impacts related to habitat loss and fragmentation. Further, a review of Environmental Impact Statements (EISs) was performed with the aim to study how impact prediction and assessment concerning biodiversity issues were addressed. Finally, different approaches to the assessment of impacts on biodiversity are discussed, together with the potential offered by the implementation of GIS-based ecological models in EIA and SEA.

2. Ecological models for prediction of fragmentation effects

Since habitat fragmentation is a problem of major concern for biodiversity, methods for quantifying and modelling effects of fragmentation will be necessary in biodiversity impact prediction. Biodiversity components and their properties, for example the requirements of vulnerable species, can be modelled at landscape and population levels, based on habitat representation in GIS. Within research on landscape ecology, spatial ecology and conservation biology, such GIS-based predictions and simulations are growing fields (e.g., Scott et al., 2002). This is due to improvements in computer hardware, GIS software, remote sensing, management of databases, and statistical modelling (Guisan and Zimmermann, 2000; Lehmann et al., 2002a). Fig. 1 presents a non-exhaustive selection of models that have potential as relevant tools for the improvement of predictions in biodiversity assessment. A general feature of all these models is the possibility to apply them at landscape and regional levels. The selected models are spatially explicit, which implies that they are or could be implemented in a GIS format and that it would be possible to adapt them to new environments and situations. One distinction made in Fig. 1 is that between habitat suitability (HS) models and metapopulation models. The former provide distribution maps of occurrence probabilities, based on habitat suitability and/or accessibility for biodiversity components, whereas the latter calculate population dynamics and viability of populations in fragmented but partly connected habitats.

Another distinction that could be made among the models presented in Fig. 1 is between expert models and models requiring empirical data. In expert models, such as LEDESS (Knol et al., 1999) and HSI (Mason et al., 1979; Hays et al., 1981), parameters are obtained from literature and/or expert opinion. Expert models may in turn rely on empirical research, but they are designed for incorporating existing parameters, not deriving them. Empirical models have parameter values that are derived from empirical data (Maurer, 2002). They can be more process-based or mechanistic, based on the underlying causal mechanisms of the problem under

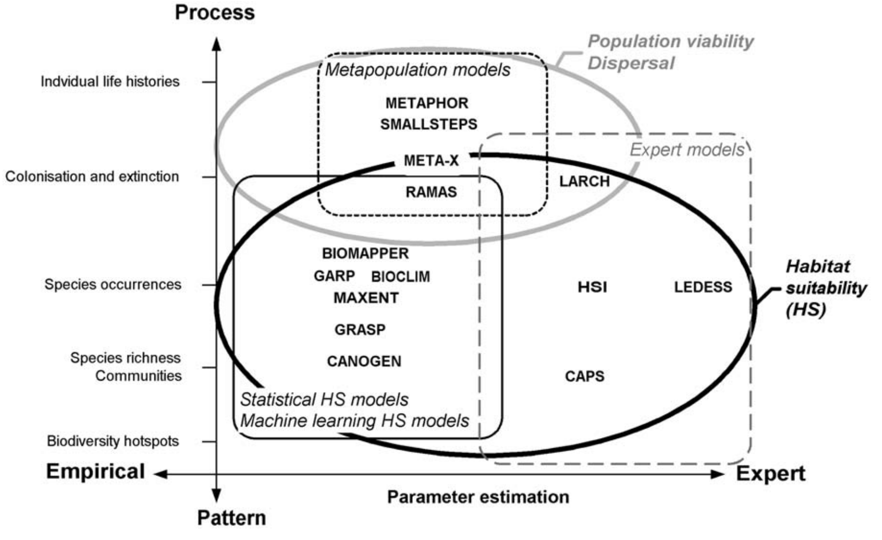

Fig. 1. Ecological models with potential as prediction tools are characterized in terms of metapopulation models, statistical models and expert models. The resulting outputs of the models are characterized as population viability or habitat suitability. See text for references.

consideration, and can thus be very detailed. Empirical models can also be more pattern-based or phenomenological, attempting to indirectly describe the relationship between habitat and for instance population dynamics, without incorporating data on the actual mechanisms that results in, for example, population change. Such pattern-based models seek to uncover relationships between biodiversity components and habitat features and patterns at the landscape scale. Correlations between populations and habitat characteristics uncovered by such techniques are static and do not necessarily represent cause–effect relationships (Guisan and Zimmermann, 2000; Maurer, 2002).

Empirical, pattern-based HS models can be based on numerous statistical methods such as bioclimatic envelopes (BIOCLIM, Busby, 1991), regression analysis (e.g., GRASP, Lehmann et al., 2002a, 2003), ordination techniques (CANOGEN, Guisan et al., 1999) and ecological niche factor analysis (BIOMAPPER, Hirzel et al., 2002). They can also be based on machine learning techniques such as MAXENT (Phillips et al., 2004) and GARP (Stockwell and Peters, 1999), the latter of which combines several algorithms (e.g., bioclimatic envelopes and logistic regression). For a review and comparison of different modelling techniques, see Guisan and Zimmermann (2000), Elith (2002), and Dettmers et al. (2002).

Many types of biodiversity components can be modelled spatially using GIS-based HS models, including vegetation types, single species occurrence and density, multiple species, species interactions, functional types of species, species richness, communities, and biodiversity hotspots (Guisan and Zimmermann, 2000; Lehmann et al., 2002a,b). Suitable habitats for population attributes such as colonisation, extinction and dispersal can also be modelled (e.g., Hanski, 1999; Opdam et al., 2002). Some of the modelling techniques, such as regression methods, are based on data on presence (or abundance) and absence of biodiversity components (e.g., GRASP), whereas other techniques can use presence-only data (BIOCLIM, GARP, BIOMAPPER). Other models are designed for incorporating data on many individual species together (e.g., CANOGEN) or communities (e.g., CAPS, McGarigal et al., 2001). The predictive environmental variables for HS models may consist of data on abiotic and biotic conditions, such as topography, climate, land-cover, vegetation and human developments, and are often derived using remote sensing technologies. Both habitat quality and spatial aspects like quantity and connectivity can be addressed.

For single species in fragmented habitats, metapopulation models (Fig. 1) can be applied. A metapopulation is a set of local populations that may exchange individuals through dispersal (Levins, 1969). Metapopulation models (e.g., METAPHOR, Verboom et al., 2001; RAMAS, Akçakaya, 2001; META-X, Grimm et al., 2004) can relate habitat to population processes, such as colonisation and extinction, and parameters needed for long term persistence of species can be estimated (Akçakaya, 2001; Opdam et al., 2002). Metapopulation models can demonstrate that stability may occur at the landscape level, in spite of instability at the local level (Levins, 1969). They can also demonstrate the significance of dispersal in fragmented landscapes: species may become extinct in the presence of suitable habitat if the rate of colonisation is too weak to compensate for the rate of local extinction (Hanski, 1999; Akçakaya, 2001; Verboom et al., 2001). Dispersal per se for different species can also be modeled (e.g., SMALLSTEPS, Verboom et al., 2001).

Different types of models can be combined, such as landscape simulation, HS, metapopulation and dispersal models (Akçakaya, 2001; Gross and DeAngelis, 2002; Larson et al., 2004; Mörtberg and Karlström, 2005). More detailed models can be linked to provide, for instance, expert models with knowledge derived from empirical research, something that may be necessary for informed planning and decision-making. One such example is LARCH, an expert model for decision support, where a combination of expert judgements and simulations, from for example the metapopulation model METAPHOR, provide input to general planning rules for species persistence in different landscapes (Verboom et al., 2001).

When habitat variables are obtained from GIS-based environmental information, and the biodiversity component's multiple response (i.e., its ecological profile) is derived through any of the described modelling techniques, or otherwise known (and implemented into expert models), the potential distribution of such a biodiversity component can be predicted within the modelled area (Guisan and Zimmermann, 2000). Further, the creation and alteration of landscape scenarios in GIS makes it possible to predict and assess the impacts of planned developments and landscape changes on biodiversity components (e.g., Harms et al., 2000; Mörtberg, 2004). This means that impacts of, for instance, habitat fragmentation can be quantified and visualised, and alternative planning scenarios can be compared and evaluated from a biodiversity perspective.

3. Review of environmental impact statements

As part of the research project, a review study was conducted on EISs addressing road and railway projects, from four countries that are members of the European Union (EU). The objectives of the review were to study the current state of integration of biodiversity issues, the use of prediction methods and the considerations of effects of habitat loss and fragmentation in biodiversity assessments. The aim was to identify knowledge gaps and needs for improvements.

3.1. Review methods

A total of 38 EISs from Sweden, France, the UK and Ireland were reviewed. As all reports were from EU countries, they share an EIA legislation that complies with the EU directive on EIA (OJ, 1985, 1997). The temporal variation in the implementation of EIA regulations combined with practices and specific traditions on nature protection imply dissimilarities in the way the EIA process is perceived and applied in these countries. The EISs were selected based on the following criteria: the projects should have been carried out in EU member countries (limited to Swedish, French and English speaking countries), they should concern roads and railways, the reports should be published after 1999[3], and the projects should have a spatial scale larger than a kilometre. The number of EISs collected for the different countries varied depending on the difficulties to gather documents fulfilling the named requirements.

Since all the studied EISs concerned road and railway projects, similar types of impacts on biodiversity and similar methodologies could be expected. The review was conducted systematically, following a review checklist consisting of 12 questions with multiple-choice answers. The review was partly based on content analysis methodologies according to Krippendorff (1980). The main focus of the checklist was to characterise the terminology, data and methodology that had been used concerning biodiversity issues.

One theme of questions concerned whether or not biodiversity was taken into account in the EIS. This included the use of the term biodiversity, which was classified into four different categories depending on whether it occurred in the title of the chapter concerning such issues, in the aim and objectives section, in the description of the environment or as a reference to policy measures and other documents. The consideration of different types of impacts on biodiversity related to infrastructure, such as fragmentation and barrier effects, was studied. A second theme of review questions concerned methods used for the biodiversity assessments, such as species inventories/lists, prediction methods, GIS-based models, cartographic material, etc. Information on references to Natura 2000 areas (OJ, 1979, 1992), protected areas and protected/red-listed species was also extracted. Information on the type of assessment was collected focusing on the use of qualitative or quantitative information and on the distinction between different steps in the impact assessment. By quantitative methods it was meant any attempt to quantify the biodiversity resources of the area in question or the potential loss of these resources as a result of the project. Further, we studied whether the results of the impact predictions (using specific methods or not) were used to make impact evaluations and study the consequences of the predicted impacts, and whether mitigation measures dealing with biodiversity issues were mentioned in the EIS.

A third theme of questions dealt with levels of biodiversity and spatial scale, which is closely related to effects of habitat loss and fragmentation. The vocabulary used to characterise the environment, such as species, habitat and/or biotope, population, landscape or ecosystem, was analysed in order to identify what approaches were used to study the state of the environment and the impacts of the projects on biodiversity. It also provided information on what geographical scale was used to describe the natural environment. For instance, the species approach corresponded to reports that considered the presence/absence of species without looking at their population or the distribution of their habitat. The next theme of questions concerned the integration of a time scale when dealing with impacts on biodiversity. The study of time considerations in the biodiversity assessment was subdivided into three points: the consideration of impacts for both construction and operation phases, the distinction between short-term and long-term impacts, and the presence of information on monitoring of ecological parameters.

[3] Corresponds to the implementation of the EU directive in the Swedish legislation.

3.2. Review results

While performing the review, no information on biodiversity was found in the EU mandated section that deals with landscape. The landscape assessment within the EIA process was restricted to aesthetical values, sometimes divided into landscape character effects and visual effects, and therefore did not contain information on biodiversity aspects. Instead, biodiversity issues were treated in a separate section that was called “natural environment” in most reports but also “ecology and nature conservation” or “flora and fauna” in some other EISs.

Generally there were large variations in the quality of the biodiversity assessments within the EISs, even within the same country. It was possible to identify similarities specific to each country, however a high heterogeneity in the overall quality of the assessment remained. Out of the 38 EISs reviewed, 19 were from Sweden, 10 from France, five from the UK and four from Ireland. They were all published between 1999 and 2003 and the projects described in the documents ranged from 1.5 to 200 km in length (Fig. 2). It should be pointed out that information on the total length of the road or railway, though a primordial characteristic of the project, was not always specified in the EISs. This information, which relates directly to the magnitude of the impacts, was difficult to find in many cases and was missing for a handful of projects. When this information was not available, the size of the project was estimated from the cartographic material included in the report.

The term biodiversity was used in only 20 out of 38 EISs. A definition for biodiversity and what it implied in the EIA process could not be found in any of the reports. In the seven reports where biodiversity was mentioned in the section dealing with the aims of the EIS, no reference to biodiversity could be found further on in the document. There were only five EISs where biodiversity was used in more than one section and only one document used the term biodiversity in all four sections, including the title of the section dealing with impacts on biodiversity.

The results showed that fragmentation and barrier effects related to the road and railway projects were not systematically taken into consideration. Impacts linked to the barrier effects were studied or mentioned in 74% of the reports whereas only 42% of the projects considered fragmentation related issues. In many EISs, the impact assessment often remained on a descriptive level and therefore considered only direct impacts, such as local habitat loss for some species, without considering indirect impacts linked to the overall habitat fragmentation on a landscape level.

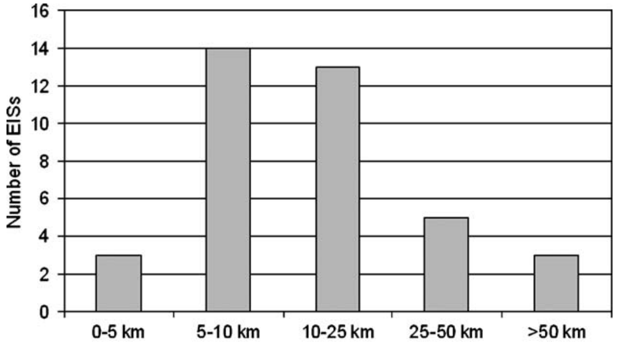

Fig. 2. Length of the road and railway projects in reviewed EISs.

All reports except one contained information on areas having some level of nature protection. References to the Habitat directive and Natura 2000 network were found in 17 reports from all four countries. A qualitative assessment was conducted in all reports but a quantification of some of the impacts on biodiversity was produced for only eight projects. This resulted in descriptive assessments rather than analytical and predictive. Concerning the distinction between different steps in the assessment of each type of impact, the vocabulary or concepts used varied among countries. In Sweden, the SNRA (2002) recommends differentiating effects and consequences, while the EISs from the UK and Ireland distinguished between impact magnitude and impact significance. No attempt to present different steps was found in the EISs from France. Of the 38 EISs that were reviewed, only five considered different steps to describe and assess impacts on biodiversity.

Details on the methods that were used or the fieldwork that was performed for the biodiversity assessment were seldom presented in the reports (Fig. 3). In 50% of the reports, it was either impossible to determine what methodology had been used for the biodiversity assessment or no specific methodology was used. Species inventories were the main methods for biodiversity assessment, and 45% of the EISs reported on species inventories that had been performed or were available for the study area. In 31% of the reports, methods other than species inventories were used to perform the biodiversity assessment. These included, among others, habitat surveys, the use of indicator species, use of historical information, the elaboration of ecological resources value criteria, and the use of species presence/abundance coefficient methods. Specific maps presented either in the baseline or the impact assessment showing the ecological resources or localising impacts on biodiversity were available in 32 projects, while six failed to provide such cartographic information. The quality of the cartographic data, when provided, varied dramatically between projects. The use of GIS in the biodiversity assessment was restricted to display and mapping functions whereas analytical capacities of a GIS were not used.

The vocabulary used in the assessment can provide information on the type of approach employed and the ecological or geographical scale used in the assessment. The most common approach was based on studies at the local habitat level (habitat and biotope), as shown in Fig. 4. The term biotope, the meaning of which is similar to habitat, was included in the study due to its use in EISs from both France and Sweden, though these countries also use the term habitat.

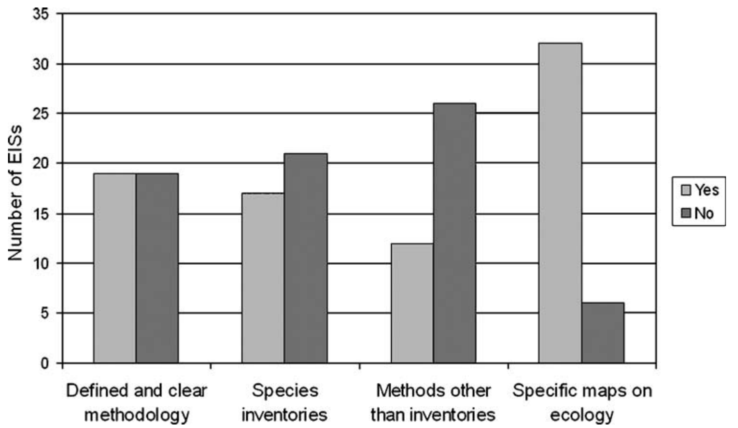

Fig. 3. Methodology used in the biodiversity assessment.

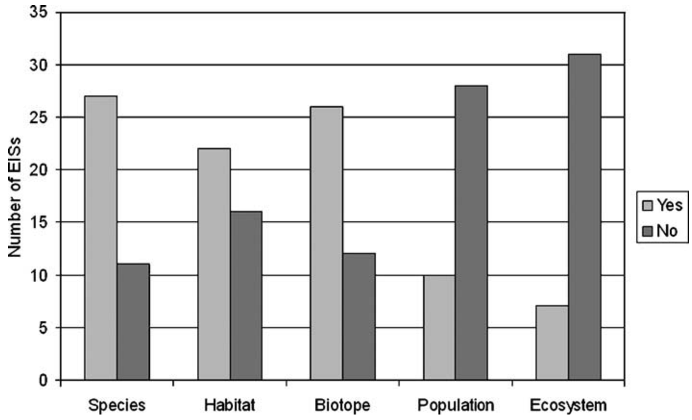

Fig. 4. Frequency of the use of key ecological terms in the biodiversity assessment.

Studies using species data were also common and could be found in 71% of the reports, whereas considerations of population or ecosystem can be found in 26% and 18% of the reports, respectively. This implied that studies within the biodiversity assessment often concentrated on the local scale and rarely looked at impacts on the ecosystem level, or at landscape or regional scales to describe or predict impacts on biodiversity. Species considerations often stopped at the identification of the presence/absence of protected or red-listed species without considering the population density or distribution, and multi-species inventories were rare in the EISs. Considering this, the local habitat considerations were the most widespread for biodiversity assessment in the EISs.

Most EISs considered a time perspective in the assessment. Fig. 5 shows that nearly all EISs considered impacts during both the construction and operation phases of the project. But impact assessment during the construction phase was often limited to a standardised section of potential effects occurring during construction, avoiding dealing with the specificities of the project. A distinction between long-term and short-term impacts was seldom made, while most of the assessments dealt with short-term impacts. Long-term impacts were considered in some cases, but not explicitly or in a specific chapter. It should be noted that in one EIS, a study on the

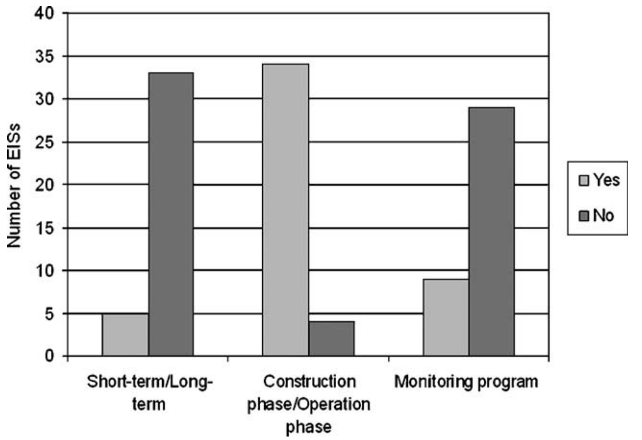

Fig. 5. Consideration of issues related to the time scale in the assessment.

natural vegetation succession was initiated to analyse the ecosystem dynamic and the ecological value of a specific area over a 20-year period. The use of a monitoring program in the EIA process, to assess the accuracy of the impact predictions, can be seen as another indication of the consideration of a time perspective. In the EISs that were reviewed, only 24% provided any information about the monitoring of some potential impacts. The monitoring programs proposed in the reports varied from standard instructions for follow-up studies to specific recommendations on parameters to be measured after implementation of the project. It should however be noted that the inclusion of a monitoring program is not a requirement in the EIA directive (OJ, 1985). Moreover, depending on the road or railway planning system in the different countries, a monitoring program might be proposed at a later stage of the planning process and may therefore not be included in the EIS.

4. Biodiversity impact prediction in EIA

4.1. Current practice

Even though some of the studied EISs reached high quality on specific points, the results of the review show that despite the EU directive on EIA being enacted over 20 years ago and recent efforts regarding biodiversity issues (e.g., the ratification of the CBD), the assessment of biodiversity related impacts in the EIA process is still far from meeting its goals. This lack of consistent quality in biodiversity assessment has been repeatedly pointed out in literature (Treweek et al., 1993; Thompson et al., 1997; Byron et al., 2000; Atkinson et al., 2000). The review showed that specific impacts on biodiversity, such as defined by the CBD, are not yet or very rarely considered. Even though the biodiversity concept is now part of the scope of the EIA process according to the requirements of the CBD (Slootweg and Kolhoff, 2003), it is still neglected in many projects. A probable explanation is that biodiversity remains abstract to practitioners and that biodiversity assessment implies the use of methodologies that are not part of current EIA routines. This might depend on competence problems and/or the lack of adequate methodologies. Biodiversity assessment needs specific methods to assess impacts on biodiversity that provide relevant and reliable predictions for the EIA process. The different levels of biodiversity (genetic, species and ecosystem) often appeared to be neglected in the EIA process. However, the integration of biodiversity in the scope of the EIA process is rather recent, and a few years of inertia in the implementation of new practices concerning biodiversity can be expected.

According to the results of the review, most EISs produced today, to some extent, considered the species and habitats components, even though they were often restricted to protected areas and protected species, and seldom considered the ecosystem level. This confirms the results of Byron et al. (2000), who identified an emphasis being placed on formally protected sites and protected species in biodiversity assessments. However, the practice of nature conservation planning has generally not been systematic and protected areas have been established for a diversity of reasons, including recreational and scenic values (Margules and Pressey, 2000). But, according to these authors, protected areas are increasingly being established principally for the protection of biodiversity, taking into account representativeness and persistence of natural processes and viable populations. For instance, the ecological network Natura 2000 (OJ, 1992) was created for the long-term protection of rare and endangered species and natural habitats. Thereby a more quantitative and spatially explicit approach to conservation planning can be achieved.

According to the review, Natura 2000 was frequently referred to in the EISs, and provided information on the importance of sites and species in an area, in a regional, national or European perspective, and thereby contributed to the assessment of impacts on biodiversity. However, even though the consideration of Natura 2000 and other protected areas is necessary, it might not be enough to fulfil the ambitions and requirements of the EIA process on biodiversity issues.

Protected areas alone may not be adequate for protecting all aspects of biodiversity, particularly in productive landscapes and landscapes with development potential (Margules and Pressey, 2000). For instance, local impacts on non-protected species that may be judged as insignificant when considered in isolation, may become significant if replicated over large areas as a result of other developments (Treweek et al., 1998). The lack of knowledge about areas and species that do not benefit from a protection status is problematic, since they may fulfil important functions in the ecosystem or landscape, which in the long-term might result in serious adverse effects on biodiversity and on surrounding protected areas. Protected areas and species may be prioritised not only because of their significance, but also because data are readily available, whereas for non-protected areas time consuming and costly data collection needs to be undertaken.

In many EISs, the biodiversity assessment remained on a descriptive level and therefore often considered only direct impacts, for example local habitat loss for certain species due to land being developed, without considering indirect, long-term, cumulative or widespread impacts. Byron et al. (2000) previously identified the absence of information on cumulative impacts as well as the lack of information on indirect effects. Concerning the type of impacts that were presented in the assessments, even well known impacts linked to linear projects, such as fragmentation and barrier effects (e.g., Trocmé et al., 2002), are often not considered in the biodiversity assessment.

The descriptive nature of biodiversity assessments is directly linked to the lack of quantification and prediction of potential impacts. The results of the review showed that, despite the advances in predictive ecological modelling, prediction tools were not used in biodiversity assessment. Likewise, Piepers et al. (2002) pointed out that the implementation of predictive modelling to assess fragmentation effects is still at an early stage of development. One reason for the lack of impact quantification and prediction could be the absence of ready-to-use, straightforward methodologies (Thompson et al., 1997), which would allow predictions and comparisons to be made with the current state of the environment.

4.2. Potential prediction tools for biodiversity assessment

A wide range of ecological models can potentially be used as prediction tools for biodiversity assessments in the EIA and SEA processes. Selecting which models to use will depend on the aim and scope of the study, and the context in which the results will be used. Further issues that need to be considered are, for instance, what biodiversity components are to be modelled, availability and quality of data and expert knowledge, time frame, available resources, and competence of those carrying out the analyses. In addition, the limitations and constraints of the different types of ecological models must be taken into account.

Since process-based models are based on cause–effect relationships, they are likely to provide more accurate predictions under a wide range of conditions, including outside the area and time-frame for which they were constructed. On the other hand, they are very data-intensive and need a large number of parameters (Maurer, 2002). Metapopulation models investigate the potential for identified habitat to be colonised by specific species and assesses the development and viability of the population (Akçakaya, 2001). Thus, they provide a good potential for assessing long-term effects of habitat loss and isolation on the persistence of species. They are confined to the assessment of single species, though, and require more parameters than HS models.

Pattern-based models, such as statistical HS models, are generally less data-intensive and can be constructed to provide predictions across large spatial and temporal scales (Guisan and Zimmermann, 2000; Maurer, 2002; Lehmann et al., 2002a). There are many different techniques for constructing HS models based on empirical data (Fig. 1). Several of these techniques have been tested and compared by, for example, Elith (2002) and Dettmers et al. (2002). According to these authors, a general conclusion is that data quality is more critical than which model is chosen. Consequently, when decisions are to be based on models containing relatively large degrees of uncertainty, caused by, for instance, scarce, inaccurate or unrepresentative data, many different modelling techniques can be applied and compared (Dettmers et al., 2002; Zabel et al., 2002).

An advantage using empirical data in HS-models is that the models are derived directly from the data, which imply that the models are locally relevant. Some disadvantages of HS-models are that they depend on the availability of data and they are static in time (Guisan and Zimmermann, 2000). Further, even if validation of the model with an independent dataset is implicit in such methods, habitat associations of a given species or biodiversity component can vary, for example, over time, across regions, and across different population densities (Boone and Krohn, 2002). Thus, since pattern-based models do not necessarily represent causal effects, the results can only be generalised to make predictions within the environmental conditions found in the data used to construct the models (Lehmann et al., 2002a). This means that extrapolation outside the range of parameter values of the specific situation the model was tuned for, can give misleading results.

Expert models can differ widely, and the availability and state of expert knowledge is obviously crucial. Expert models that aggregate scientific knowledge, both through expert knowledge and the linking of more detailed models, can provide parameters for the persistence of populations in a landscape (Vos et al., 2001). This may solve problems caused by, for instance, a lack of distribution data, occasionally unoccupied but suitable habitats, and time lags in population response to landscape change (Opdam and Wiens, 2002). Constructing large expert models and databases, which can provide parameters derived from detailed research on the problems of concern, may be attractive particularly if all high priority biodiversity components are incorporated. However, this is not always possible to achieve. Maurer (2002) stated that when ecological models were used for decision-making there was a preponderance for statistical models, which, according to the author, probably reflected the reality of conservation decision making with limited resources. Metapopulation and other more detailed models were, according to the same author, most often associated with species of economic (game) or legal (endangered) importance.

The use of GIS-based ecological models as prediction tools has certain limitations, which have to be considered (e.g., Scott et al., 2002). For instance, species and other biodiversity components are not equally likely to be successfully modelled, for reasons such as differences in habitat specificity and in how well remotely sensed data match habitat needs. Further, there is a lack of knowledge on the actual response of biodiversity components to infrastructure and other developments (e.g., Piepers et al., 2002). Biotic interactions, disturbance events and ecological processes may not yet be addressed properly by current methods.

Still, despite the limitations, several authors now claim that GIS-based predictive models have reached a level of quality such that they may be considered as valuable planning and assessment tools (e.g., Guisan and Zimmermann, 2000; Lehmann et al., 2002a; Johnson et al., 2004). In particular, they can deliver a base for quantitative predictions when spatially explicit information is needed. In addition, biodiversity assessment depends on data, for example, on the distribution of habitats and species. Reliance on imprecise estimates of species distributions, as provided by patchily distributed point data from field surveys or biological collections, has been a constraint to biodiversity studies and assessments (Treweek et al., 1998; Lehmann et al., 2002b). GIS-based predictive modelling is a means of making more effective use of sparse biological data by linking these data to remotely mapped environmental variables using different modelling techniques (Ferrier et al., 2002a,b).

Uncertainty may increase though when using available data. An estimation of uncertainty should be included in EIA/SEA (Treweek, 1996; Geneletti et al., 2003). In the reviewed GIS-based models, results from the models involve some degree of uncertainty; however, those uncertainties can be measured and mapped (e.g., Elith et al., 2002). Further, when results are highly uncertain, even if the absolute figures in the results should be handled with caution, they still may provide robust results when comparing planning scenarios against one another (Guisan and Zimmermann, 2000).

When impacts can be quantified and visualised, different landscape scenarios can be compared and evaluated from a biodiversity perspective. Further, vegetation succession can be modelled on a landscape level in order to consider long-term effects (e.g., in forest landscapes, Mladenoff, 2004), and related to other biodiversity components, for instance habitat suitability for animal species. Today, GIS-based habitat models are increasingly being used, for example, in the design of ecological networks at large spatial scales (Ministerie LNV, 1990; Bani et al., 2002), for forest management (Angelstam et al., 2005) and for strategic conservation planning (Margules and Pressey, 2000). In addition, Mörtberg (2004) developed a method for landscape ecological assessment in peri-urban areas, which included the formulation of regionally relevant biodiversity targets, indicator selection, predictive modelling, assessment and the possibility to iterate planning scenarios.

4.3. Approaches to biodiversity assessment

Different approaches to biodiversity assessment are summarised and illustrated in Fig. 6. The distinction between different approaches has been derived from current practices, recommendations and guidelines on biodiversity/ecological assessment and the potential contribution of GIS-based ecological models to perform predictions.

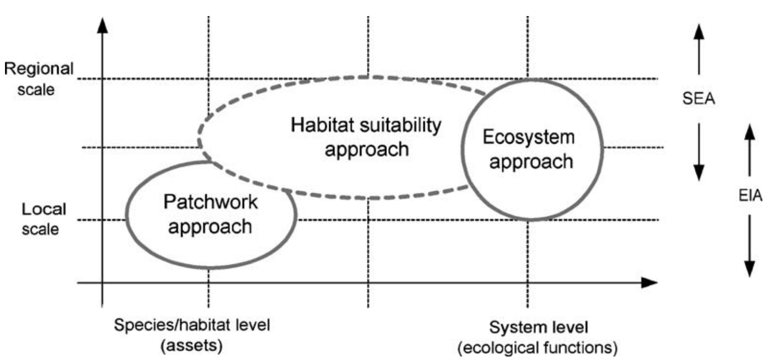

Fig. 6. Summary diagram of potential methodological approaches to biodiversity assessment, and their relationship to physical scales and ecological levels.

A first approach to biodiversity assessment could be called the patchwork approach. This approach, common to all four counties in the review, reflects some of the current practices in biodiversity assessment in the EIA process, where single sites, protected areas and protected species are focused upon. According to the review, most EISs contained an assessment of isolated features possessing a biodiversity value, whose integration in the assessment was guided by regulations on nature protection (species and areas). The patchwork approach includes both species and habitat levels, but results in an assessment done on a patch by patch basis, in one localised project at a time and in the absence of a general overview. The patchwork approach results in an assessment where the scale of ecological processes is not considered.

A second approach could be presented as the ecosystem approach (Fig. 6). This is a holistic and most of all functional and dynamic strategy that also takes into consideration the interactions between the components of an ecosystem. Petit et al. (1996) stated that important properties of an ecosystem are based on the sum of the interactions between species and not on maintaining the integrity of the species list from that ecosystem. This is also stressed in the French guidelines on biodiversity/ecological assessment (DIREN, 2002). The ecosystem approach involves the consideration of the environmental functions necessary to reach a sustainable development, and within the EIA process it has been proposed and promoted at the international level (e.g., CEQ, 1993; CBD, 2004). However, as shown by the review of EISs in the transport sector, biodiversity assessment seldom includes impact studies at the ecosystem level. Consequently, the application of the ecosystem approach needs to be further developed and tested, as it appears to be difficult to operationalise.

One way to overcome the difficulties of a holistic approach to biodiversity assessment could be to consider specific biodiversity components and processes attached to an ecosystem. A third approach could be proposed, namely the habitat suitability approach (Fig. 6), whereby habitat suitability is used in the broadest sense, taking into account habitat quality, quantity and connectivity. Moreover, habitat can then be seen as suitable not only for species' occupancy, but also for persistence of populations or communities, for dispersal, and other biodiversity entities or processes. This will make it possible to involve predictive modelling of biodiversity components and to take advantage of GIS-based ecological models as prediction tools.

As described, many biodiversity components can be modelled, yet a decision has to be made in prioritising which components would be representative for the quantification of significant impacts on biodiversity in any given planning scenario. Besides legislation, such priorities can be based on, for instance, regional and national goals for biodiversity, and/or indicators. The habitat suitability approach could be considered to be more flexible from a data requirement perspective and could take the form of studies on specific ecosystem resources or processes. It offers a higher degree of freedom to adapt to the specific ecosystem in question but also induces a higher degree of uncertainty. The approach could also facilitate the consideration and integration of widespread, long-term or cumulative impacts in the assessment. A habitat suitability approach to biodiversity assessment founded on the implementation of GIS-based ecological models as prediction tools could function as a link between the patchwork and ecosystem approaches (Fig. 6). The flexibility and limitations of the different modelling techniques should guide the selection of appropriate methods when combining studies at the local level with broader scale studies. In time, the habitat suitability approach could be seen as a step towards the ecosystem approach, which concentrates on specific functions and assets of the ecosystem concerned.

Moreover, different approaches could be combined. A similar strategy was applied by Fernandes (2000) in an EIA for a road project, where a study was performed at the landscape level in conjunction with the consideration of protected habitats and species. Considering the possibilities offered by GIS and ecological modelling, different modelling techniques could also be combined. Data on unique features and single-species modelling can be combined with modelling (mapping) of vegetation types, communities and processes at landscape and regional scales. This combination of approaches corresponds to a fine/coarse filter strategy (Noss et al., 1999; Ferrier et al., 2002a,b). The modelling of unique features and priority species, such as protected, red-listed or focal species, could be seen as the fine filter, while the modelling of vegetation types, communities and processes could be seen as the coarse filter. The latter could be the only available option when data is scarce. The habitat suitability approach in a fine/coarse filter strategy is a way of making maximum use of all available information, and of incorporating larger scales in the biodiversity assessment.

However, the ecosystem and habitat suitability approaches, to some extent, reveal limitations of the EIA process. Particularly in the case of smaller projects, it could be difficult to justify studies of the impacts on biodiversity components at the ecosystem level or at regional scales. Still, the biodiversity levels and geographical scales used in the assessment are of primary importance. The landscape level could be considered as the upper level encompassing the species and ecosystem levels (Wiens, 2002). When a biodiversity assessment is applied on the level of individual projects, it often fails to ensure adequate consideration of potentially serious cumulative and synergistic ecological effects (Treweek, 1996). For instance, road projects always form part of a larger infrastructure network, where synonymous effects with other road links, or with other projects or natural barriers, may magnify the significance of the impacts (Trocmé et al., 2002). From this perspective, SEA could provide an opportunity to determine whether proposed developments, when considered in their entirety, are compatible with national or international goals for biodiversity.

5. Conclusions

The study confirms the lack of a consistent quality in current biodiversity assessments, which is far from fulfilling the ambitions of regulations and guidelines. According to the review, the term biodiversity is seldom used in today's EIA practice, and its scope and meaning are not defined. Most EISs consider species and local habitats even though they are often restricted to protected species and protected areas. However, they rarely consider the ecosystem level.

The omission of areas not benefiting from a protection status is problematic, since those still may contain biodiversity values and/or fulfil important functions in the ecosystem or landscape. The biodiversity assessment was typically confined to local scales, which did not allow prediction and assessment of effects of habitat loss and fragmentation, nor the consideration of scales of ecological processes. A major problem that remains is the descriptive nature of many assessments, and the lack of quantifications and methods for impact predictions. Thus, the development and implementation of new methods appear necessary to meet regulations and recommendations on the consideration of biodiversity in EIA and SEA.

The use of GIS-based ecological models has potential to address several shortcomings of today's biodiversity assessment. Such models can be applied over large areas, making it possible to quantify impacts, to model and visualise uncertainty, to make better use of scarce data, and to take into account wide-spread, off-site and long-term effects. Thereby, the spatial and temporal scales of ecological processes can be taken into account, and impacts of changes such as habitat fragmentation can be quantified and predicted. Further, modelling of coarse-scale and more specific biodiversity components can be combined. Requirements and limitations of the models have to be taken into account, considering for instance data, expert knowledge and resources. GIS-based ecological models continue to have potential as prediction tools for biodiversity assessment, providing a quantitative approach and allowing impact predictions to be made not only for the study area itself, but also for the surrounding environment.

The literature and EIS reviews resulted in the identification of different approaches to address biodiversity issues in the EIA process. The patchwork approach illustrates current practices, where the assessment concentrates on single sites, protected areas and protected species. However, recent guidelines and recommendations advocate for an ecosystem approach to biodiversity assessment. The habitat suitability approach, using GIS-based ecological models, offer potential for assessing fragmentation problems. This approach could bridge the gap between expressed ambitions and current practice, and allow for quantified predictions and more systematic biodiversity assessments. Moreover, the consideration of biodiversity requires a holistic approach where up-scaling from the local level to the ecosystem or landscape levels is a necessity. While the application of these different approaches might not always fit within the ambits of the EIA process, the SEA process could offer a better framework for predicting impacts at landscape level. In this way, major threats like habitat fragmentation and degradation could be addressed and a more sustainable development could be promoted.

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References

Akçakaya HR. Linking population-level risk assessment with landscape and habitat models. Sci Total Environ 2001;274:283–91.

Angelstam P, Roberge J-M, Löhmus A, Bergamis M, Brazaitis G, Breuss M, et al. Habitat modelling as a tool for landscape-scale conservation – a review of parameters for focal forest birds. Ecol Bull 2005;51:427–54.

Atkinson SF, Bhatia S, Schoolmaster FA, Waller WT. Treatment of biodiversity impacts in a sample of US environmental impact statements. Impact Assess Proj Apprais 2000;18:271–82.

Bani L, Baietto M, Bottoni L, Massa R. The use of focal species in designing a habitat network for a lowland area of Lombardy, Italy. Conserv Biol 2002;16:826–31.

Boone RB, Krohn WB. Modeling tools and accuracy assessment. In: Scott JM, Heglund PJ, Morrison ML, Haufler JB, Raphael MG, Wall WA, Samson FB, editors. Predicting species occurrences: issues of accuracy and scale. Washington: Island Press; 2002. p. 265–70.

Busby JR. BIOCLIM – a bioclimatic analysis and prediction system. In: Margules CR, Austin MP, editors. Nature conservation: cost effective biological surveys and data analysis. Canberra: CSIRO; 1991. p. 64–8.

Byron HJ, Treweek JR, Sheate WR, Thompson S. Road developments in the UK: an analysis of ecological assessment in environmental impact statements produced between 1993 and 1997. J Environ Plan Manag 2000;43:71–97.

Canadian Environmental Assessment Agency (CEAA) S. A guide on biodiversity and environmental assessment. Canada: Ministry of Supply and Services; 1996. 15 pp.

Convention on Biological Diversity. Decision VII/11 Ecosystem approach, URL: http://www.biodiv.org/decisions/default.aspx?m=COP-07&id=7748&lg=0; 2004. Latest accessed in March 2005.

Council of Europe. Convention on the conservation of European wildlife and nature habitats Bern 19IX. Eur Treaty Ser 1979;104.

Council on Environmental Quality (CEQ). Incorporating biodiversity considerations into environmental impact analysis under the national environmental policy act. Washington DC: Council on Environmental Quality; 1993. 29 pp.

Department of Transport (DoT). Design manual for roads and bridges-DMRB. Environmental Assessment, vol. 11. London: HMSO; 1993.

Dettmers R, Buehler DA, Barlett JB. A test and comparison of wildlife-habitat modelling techniques for predicting bird occurrence at a regional scale. In: Scott JM, Heglund PJ, Morrison ML, Haufler JB, Raphael MG, Wall WA, Samson FB, editors. Predicting species occurrences: issues of accuracy and scale. Washington: Island Press; 2002. p. 607–15.

Direction régionale de l'environnement de Midi-Pyrénées. Guide sur la prise en compte des milieux naturels dans les études d'impact. 75 pp.

Elith J. Quantitative methods for modeling species habitat: comparative performance and an application to Australian plants. In: Ferson S, Burgman M, editors. Quantitative methods for conservation biology. New York: Springer; 2002. p. 39–58.

Elith J, Burgman MA, Regan HM. Mapping epistemic uncertainties and vague concepts in predictions of species distributions. Ecol Model 2002;157:313–29.

European Commission. Report from the Commission to the European Parliament and the Council on the Application and Effectiveness of the EIA Directive (Directive 85/337/EEC as amended by Directive 97/11/EC) How successful are the Member States in implementing the EIA Directive.

Fahrig L. Relative effects of habitat loss and fragmentation on population extinction. J Wildl Manage 1997;61:603–10.

Fernandes JP. Landscape ecology and conservation management – evaluation of alternatives in a highway EIA process. Environ Impact Assess Rev 2000;20:665–80.

Ferrier S, Watson G, Pearce J, Drielsma M. Extended statistical approaches to modelling spatial pattern in biodiversity in northeast New South Wales: I. Species-level modelling. Biodivers Conserv 2002a;11:2275–307.

Ferrier S, Drielsma M, Manion G, Watson G. Extended statistical approaches to modelling spatial pattern in biodiversity in northeast New South Wales: II. Community-level modelling. Biodivers Conserv 2002b;11:2309–38.

Forman RTT. Estimate of the area affected ecologically by the road system in the United States. Conserv Biol 2000;14:31–5.

Geneletti D. Ecological evaluation for environmental impact assessment. Neth Geogr Stud (NGS 301). Utrecht: Knag; 2002. 224 pp.

Geneletti D, Beinat E, Chung CF, Fabbri AG, Scholten HJ. Accounting for uncertainty factors in biodiversity impact assessment: lessons from a case study. Environ Impact Assess Rev 2003;23:471–87.

Grimm V, Lorek H, Finke J, Köster F, Malachinski M, Sonnenschein M, et al. Meta-X: generic software for metapopulation viability analysis. Biodivers Conserv 2004;13:165–88.

Gross LJ, DeAngelis DL. Multimodeling: new approaches for linking ecological models. In: Scott JM, Heglund PJ, Morrison ML, Haufler JB, Raphael MG, Wall WA, Samson FB, editors. Predicting species occurrences: issues of accuracy and scale. Washington: Island Press; 2002. p. 467–74.

Guisan A, Zimmermann NE. Predictive habitat distribution models in ecology. Ecol Model 2000;135:147–86.

Guisan A, Weiss SB, Weiss AD. GLM versus CCA spatial modeling of plant species distribution. Plant Ecol 1999;143:107–22.

Gutzwiller KJ, editor. Applying landscape ecology in biological conservation. New York: Springer; 2002. 518 pp.

Hanski I. A practical model of metapopulation dynamics. J Anim Ecol 1994;63:151–62.

Hanski I. Habitat connectivity, habitat continuity, and metapopulations in dynamic landscapes. Oikos 1999;87:209–19.

Harms WB, Knol WC, Roos-Klein Lankhorst J. Modelling landscape changes in the Netherlands: the Central City Belt case study. In: Mander Ü, Jongman R, editors. Landscape perspectives of land use changes. Advances Ecol Sciences, vol. 6. Southampton (UK): WIT; 2000. p. 1–17.

Hays RL, Summers C, Seitz W. Estimating wildlife habitat variables. USDA fish and wildlife service FWS/OBS-81/47; 1981. 111 pp.

Hirzel AH, Hausser J, Chessel D, Perrin N. Ecological-niche factor analysis: how to compute habitat-suitability maps without absence data? Ecology 2002;83:2027–36.

International Association for Impact Assessment. Proposed conceptual and procedural framework for the integration of biological diversity considerations with national systems for impact assessment. Utrecht: Commissie voor de Milieueffectrapportage; 2001. 46 pp.

Johnson CJ, Seip DR, Boyce MS. A quantitative approach to conservation planning: using resource selection functions to map the distribution of mountain caribou at multiple spatial scales. J Appl Ecol 2004;41:238–51.

Knol WC, van Eupen M, Roos-Klein-Lankhorst J, Verweij P. Landscape Ecological and Evaluation Support System LEDESS. GIS Application Manual version 2.0. Technical Manual. Wageningen: Winand Staring Centre; 1999. 112 pp.

Krippendorff K. Content analysis. Beverly Hills, California: Sage Publication; 1980. 191 pp.

Larson MA, Thompson III FR, Millspaugh JJ, Dijak WD, Shifley SR. Linking population viability, habitat suitability, and landscape simulation models for conservation planning. Ecol Model 2004;180:103–18.

Lehmann A, Overton JM, Austin MP. Regression models for spatial prediction: their role for biodiversity and conservation. Biodivers Conserv 2002a;11:2085–92.

Lehmann A, Leathwick JR, Overton JM. Assessing New Zealand fern diversity from spatial prediction of species assemblages. Biodivers Conserv 2002b;11:2217–38.

Lehmann A, Overton JM, Leathwick JR. GRASP: generalized regression analysis and spatial prediction. Ecol Model 2003;160:165–83.

Levins R. Some demographic and genetic consequences of environmental heterogeneity for biological control. Bull Entomol Soc Am 1969;15:237–40.

Margules CR, Pressey RL. Systematic conservation planning. Nature 2000;405:243–53.

Mason WTJ, Cushwa CT, Slaski CJ, Gladwin DM. A procedure for describing fish and wildlife: coding instructions for Pennsylvania. USDI fish and wildlife service, FWS/OBS-79/19; 1979. 21 pp.

Maurer BA. Predicting distribution and abundance: thinking within and between scales. In: Scott JM, Heglund PJ, Morrison ML, Haufler JB, Raphael MG, Wall WA, Samson FB, editors. Predicting species occurrences: issues of accuracy and scale. Washington: Island Press; 2002. p. 125–32.

McGarigal K, Jackson S, Griffin C. Conservation assessment and prioritization system: an interactive decision-support system; 2001. URL: http://www.umass.edu/landeco/research/caps/caps.html.

Ministerie LNV. Het Natuurbeleidsplan: regeringsbeslissing. Ministry of agriculture, nature management and fisheries. The Hague: SDU; 1990.

Mladenoff DJ. LANDIS and forest landscape models. Ecol Model 2004;180:7–19.

Mörtberg U. Landscape ecological analysis and assessment in an urbanising environment—forest birds as biodiversity indicators. Dissertation. Stockholm: Royal Institute of Technology, Department of Land and Water Resources Engineering; 2004. 50 pp.

Mörtberg U, Karlström A. Predicting forest grouse distribution taking account of spatial autocorrelation. J Nat Conserv 2005;13:147–59.

Noss RF, Dinerstein E, Gilberg B, Gilpin M, Miller BJ, Terborgh J, et al. Core areas: where nature reigns. In: Soulé ME, Terborgh J, editors. Continental conservation: scientific foundations of regional reserve networks. Washington, DC: Island Press; 1999. p. 99–128.

Official Journal of the European Communities. Council Directive 79/409/EEC of 2 April 1979 on the conservation of wild birds. Off J 1979. L 103: [25/04/1979].

Official Journal of the European Communities. Council Directive 1985/337/EEC of 27 June 1985 on the assessment of the effects of certain public and private projects on the environment. Off J 1985. L175. [05/07/1985].

Official Journal of the European Communities. Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora. Off J 1992. L 206. [22/07/1992].

Official Journal of the European Communities. Convention on Biological Diversity. Off J 1993. L 309/1: [13/12/1993].

Official Journal of the European Communities. Council Directive 97/11/EC of 3 March 1997, amending Directive 85/337/EEC on the assessment of effects of certain public and private projects on the environment. Off J 1997;L 073. [14/03/1997].

Official Journal of European Communities. Directive 2001/42/EC of the European Parliament and the Council of the 27 June on the assessment of the effects of certain plans and programmes on the environment. Off J 2001. L 197/30 [21/07/2001].

Opdam P, Foppen R, Vos C. Bridging the gap between ecology and spatial planning in landscape ecology. Landsc Ecol 2002;16:767–79.

Opdam P, Wiens JA. Fragmentation, habitat loss and landscape management. In: Norris K, Pain DJ, editors. Conserving bird biodiversity General principles and their application. Cambridge: Cambridge University; 2002. p. 202–23.

Petit S, Burel F, Pain G. Conséquences écologiques de la fragmentation des habitats liée aux aménagements routiers et agricoles. Les méthodes d'évaluation des impacts sur le milieuxLes ingénieurs écologues; 1996. p. 19–26.

Phillips SJ, Dudík M, Schapire RE. A maximum entropy approach to species distribution modeling. Proceedings of the 21st international conference on machine learning, Banff, Canada.

Piepers A, Alvarez G, Bouwma IM, De Vries HJG, Seiler A. Minimizing fragmentation through appropriate planning. In: Trocmé M, Cahill S, de Vries JG, Farrall H, Folkeson L, Fry G, Hichs C, Peymen J, editors. COST 341 Habitat fragmentation due to transportation infrastructure The European Review. Luxemburg: Office for Official Publications of the European Communities; 2002. p. 115–28.

Scott JM, Heglund PJ, Morrison ML, Haufler JB, Raphael MG, Wall WA, Samson FB, editors. Predicting species occurrences: issues of accuracy and scale. Washington: Island Press; 2002. 868 pp.

Seiler A. Effects of infrastructure on Nature. In: Trocmé M, Cahill S, de Vries JG, Farrall H, Folkeson L, Fry G, Hichs C, Peymen J, editors. COST 341. Habitat fragmentation due to transportation infrastructure. The European Review. Luxemburg: Office for Official Publications of the European Communities; 2002. p. 31–50.

Slootweg R, Kolhoff A. A generic approach to integrate biodiversity considerations in screening and scooping for EIA. Environ Impact Assess Rev 2003;23:657–81.

Stockwell DRB, Peters DP. The GARP modelling system: problems and solutions to automated spatial prediction. Int J Geogr Inf Syst 1999;13:143–58.

Swedish National Road Administration. Handbok, Miljökonsekvensbeskrivning inom vägsektorn. Del 3 – Analys och bedömning, 2002:43; 2002. 171 pp. [In Swedish].

Swedish National Road Administration and Swedish National Rail Administration. Assessment of ecological effects from roads and railways – recommended methodology. Vägverket 1996;33.

Thompson S, Treweek JR, Thurling DJ. The ecological component of environmental impact assessment: a critical review of British environmental statements. J Environ Plan Manag 1997;40:157–71.

Treweek JR. Ecology and environmental impact assessment. J Appl Ecol 1996;33:191–9.

Treweek JR, Thompson S, Veitch N, Japp C. Ecological assessment of proposed road developments: a review of environmental statements. J Environ Plan Manag 1993;36:295–307.

Treweek JR, Hankard P, Roy DB, Arnold H, Thompson S. Scope for strategic ecological assessment of trunk-road development in England with respect to potential impacts on lowland heathland, the Dartford warbler (*Sylvia undata*) and the sand lizard (*Lacerta agilis*). J Environ Manag 1998;53:147–63.

Trocmé M, Cahill S, de Vries JG, Farrall H, Folkeson L, Fry G, Hichs C, Peymen J, editors. COST 341 Habitat fragmentation due to transportation infrastructure The European Review. Luxemburg: Office for Official Publications of the European Communities; 2002. 253 pp.

United Nations Conference on the Environment and development (UNCED). The UN convention on biological diversity (Rio, UNCED).

United Nations Educational, Scientific and Cultural Organizations (UNESCO). Convention on Wetlands of international importance especially as Waterfowl habitat. Ramsar 2 February, The convention on Wetlands text, as amended in 1982 and 1987.

United Nations Environmental Programme (UNEP). Convention on migratory species Bonn on 23 June; 1979.

Verboom J, Foppen R, Chardon P, Opdam PFM, Luttikhuizen P. Introducing the key patch approach for habitat networks with persistent populations: an example for marshland birds. Biol Conserv 2001;100:89–101.

Vos CC, Verboom J, Opdam PFM, ter Braak CJF. Towards ecologically scaled landscape indices. Am Nat 2001;183:24–41.

Wiens J. Central concepts and issues in landscape ecology. In: Gutzwiller KJ, editor. Applying landscape ecology in biological conservation. New York: Springer; 2002. p. 3–21.

Wilcove DS, Rothstein D, Dubow J, Phillips A, Losos E. Quantifying threats to imperilled species in the United States. BioScience 1998;48:607–15.

World Bank. Biodiversity and environmental assessment toolkit. Washington DC: The World Bank; 2000.

World Bank. Environmental assessment sourcebook update: biodiversity and environmental assessment. Washington DC: The World Bank; 1997. 10 pp.

Zabel CJ, Roberts LM, Mulder BS, Stauffer HB, Dunk JR, Wolcott K, et al. A collaborative approach in adaptive management at a large-landscape scale. In: Scott JM, Heglund PJ, Morrison ML, Haufler JB, Raphael MG, Wall WA, Samson FB, editors. Predicting species occurrences: issues of accuracy and scale. Washington: Island Press; 2002. p. 241–53.