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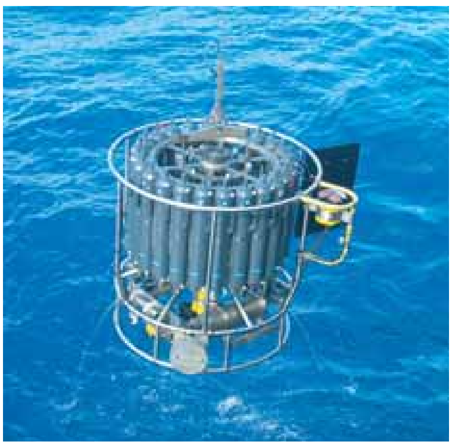
Systematic conservation planning in Europe – the case of wetland biodiversity

Kerstin Jantke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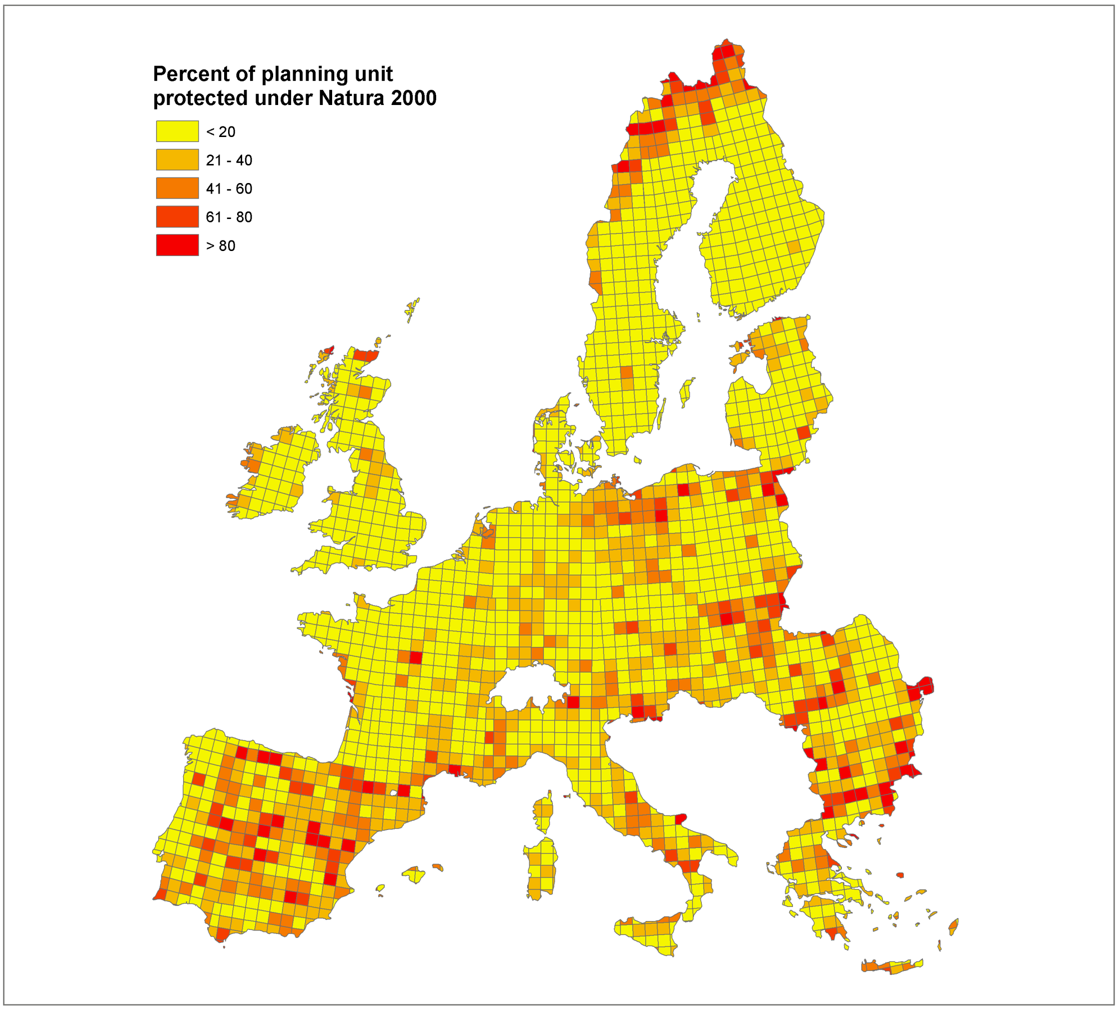

Kerstin Jantke

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Summary

Accelerated loss of biodiversity calls for effective and efficient means for the safeguarding of biodiversity features. Central to any conservation strategy throughout the world is the establishment of protected areas. The need to evaluate their effectiveness in representing and maintaining biodiversity has led to the evolvement of a sub-discipline of conservation biology called systematic conservation planning.

This thesis aims to facilitate and strengthen the application of systematic conservation planning methods to European conservation problems. It also aims at contributing to a better understanding and correct implementation of economic concepts in conservation planning applications.

Foundation of the thesis is the development of the mathematical programming model HABITAT. This reserve selection tool is based on principles of systematic conservation planning and economic theory. It is designed for the specific requirements for conservation planning on the European continent. Four papers address the application of this tool to European wetland conservation planning problems. 69 to 72 wetland dependent vertebrate species of European conservation concern serve as surrogates for biodiversity.

Starting point of the first paper called *Multiple-species conservation planning for European wetlands with different degrees of coordination* is the institutional and administrative complexity of decision-making on the establishment of protected areas in the European Union. The paper addresses the question how efficient different strategies of geopolitical coordination in conservation planning are. Results show that strong coordination reduces area requirements for conservation substantially. Furthermore, synergy effects are quantified.

The second paper, *Integrating land market feedbacks into conservation planning – a mathematical programming approach*, extends the previous model application by implementing opportunity costs for acquiring land for conservation activities. The study demonstrates a method to integrate land market feedbacks directly and consistently into conservation planning. Different cost representations are compared to illustrate the effect of incorporating the dynamic nature of opportunity costs. Results show that ignoring these feedbacks can lead to highly cost-ineffective solutions in reserve selection.

The third paper, *Benefits of global earth observation for conservation planning in the case of European wetland biodiversity*, estimates benefits of improved land cover and land value information for conservation planning. The paper presents methodologies to overcome data deficiencies by integrating available datasets from different models and sources on a European scale. Results show that the accuracy of conservation plans improves considerably with higher resolution habitat data and spatially explicit land rent data. However, the study also emphasizes the need for better resolved data on the distribution of species of European conservation concern.

The fourth paper, *Gap analysis of European wetland species: priority regions for expanding the Natura 2000 network*, extends the previous model applications by incorporating the existing system of protected areas under the Natura 2000 framework. The paper provides a systematic evaluation of the performance of the Natura 2000 system in covering endangered wetland vertebrate species. Results show that five area-demanding vertebrates are not covered adequately by the current reserve system whereas only three species are fully covered. The study furthermore identifies potentials for expanding the network to move toward complete coverage and presents spatially explicit priority regions for a cost-effective expansion.

Zusammenfassung

Der zunehmende Rückgang der Biodiversität unserer Erde erfordert effektive und effiziente Maßnahmen zum Schutz von Ökosystemen, Arten und genetischer Vielfalt. Ein zentraler Aspekt von Naturschutz-Strategien weltweit ist die Ausweisung von Schutzgebieten. ‚Systematic conservation planning', eine Teildisziplin der Naturschutzbiologie, beschäftigt sich u.a. mit der Beurteilung der Effektivität von Schutzgebieten in der Repräsentierung und langfristigen Aufrechterhaltung ihrer Biodiversität.

Die vorliegende Dissertation zielt zum einen darauf ab, diese systematischen Planungsmethoden verstärkt auch in europäischen Naturschutzfragen anzuwenden. Des Weiteren soll sie zu einem besseren Verständnis und fachlich korrekter Einbindung ökonomischer Konzepte in die Naturschutzplanung beitragen.

Grundlage dieser Dissertation ist die Entwicklung des mathematischen Optimierungsmodells HABITAT. Dieses Modell zur Schutzgebietsplanung basiert auf den Grundsätzen von ‚systematic conservation planning' und ökonomischer Theorie. Es ist explizit für die besonderen Anforderungen an die Naturschutzplanung auf dem europäischen Kontinent konzipiert. Anhand von vier Studien werden mit Hilfe des HABITAT Modells Planungsaspekte des Schutzes terrestrischer Feuchtgebiete analysiert. Je nach Studie dienen hierbei 69, 70 bzw. 72 Wirbeltierarten, die auf Feuchtgebietslebensräume angewiesen sind, als Stellvertreter für die Biodiversität der Feuchtgebiete.

Ausgangspunkt für das erste Kapitel, *Multiple-species conservation planning for European wetlands with different degrees of coordination*, ist die institutionelle und administrative Komplexität von Entscheidungen über die Einrichtung von Schutzgebieten in der Europäischen Union. Die Studie beschäftigt sich mit der Frage, wie effizient verschiedene Strategien geopolitischer Koordinierung in der Naturschutzplanung sind. Die Ergebnisse zeigen, dass starke Koordinierung der Planung mit einem deutlichen Flächeneinsparungspotential verbunden ist. Synergieeffekte, die durch übergreifende Planung entstehen, werden ebenfalls quantifiziert.

Im zweiten Kapitel, *Integrating land market feedbacks into conservation planning – a mathematical programming approach*, wird das Modell dahingehend erweitert, dass Opportunitätskosten für den Erwerb von Landflächen zum Zwecke der Unterschutzstellung einbezogen werden. Diese Studie berücksichtigt, dass die Unterschutzstellung von Gebieten

dazu führen kann, dass sich Landpreise an das veränderte Nachfrageniveau anpassen. Es wird eine Methode entwickelt, diese Rückkopplungen direkt und konsistent in die Naturschutzplanung einzubeziehen. Um den Effekt der Einbeziehung dieses Aspektes zu zeigen, werden verschiedene Darstellungen der Landkosten verglichen. Die Ergebnisse verdeutlichen, dass die Vernachlässigung der Anpassung der Landpreise in der Naturschutzplanung zu aus Kostengesichtspunkten suboptimalen Schutzgebietssystemen führen kann.

Das dritte Kapitel, *Benefits of global earth observation for conservation planning in the case of European wetland biodiversity*, beschäftigt sich damit, den Nutzen von verbesserten räumlichen Informationen über Landbedeckung sowie Landpreise für die Naturschutzplanung abzuschätzen. Die Studie stellt Methoden vor, fehlende räumliche Daten aus der Integration verschiedener Datensätze aus vorhandenen Datenquellen und Modellen zu berechnen. Die Studie zeigt, dass die Genauigkeit von Naturschutzplänen durch hoch aufgelöste Daten über die Verteilung von Feuchtgebietslebensräumen und räumlich detaillierte Landkosten deutlich verbessert wird. Jedoch wird auch die Notwendigkeit hervorgehoben, besser aufgelöste Daten über die Verbreitung von Arten zu erhalten, die von erheblichem Interesse für den europäischen Naturschutz sind.

Im vierten Kapitel, *Gap analysis of European wetland species: priority regions for expanding the Natura 2000 network*, wird das in der Europäischen Union aktuell bestehende Schutzgebietssystem Natura 2000 in die Modellanalysen einbezogen. Es wird systematisch untersucht, wie leistungsfähig die Natura 2000-Gebiete darin sind, gefährdete Feuchtgebietsarten nachhaltig zu schützen. Die Ergebnisse zeigen, dass fünf Wirbeltierarten mit großem Flächenbedarf im jetzigen Schutzgebietssystem nicht ausreichend geschützt sind, während die europäischen Vorkommen von lediglich drei weiteren Arten vollständig innerhalb von Natura 2000-Gebieten liegen. Es werden darüber hinaus Flächenbedarf und Kosten für eine mögliche Erweiterung des Schutzgebietssystems ermittelt und entsprechende Gebiete räumlich dargestellt.

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Abbreviations

EU	European Union
GAMS	General Algebraic Modeling System
GAP	Gap Analysis Program
GEO	global earth observation
GEOSS	Global Earth Observation System of Systems
GIS	geographic information system
HRU	Homogenous Response Unit
MCA	minimum critical area
MVP	minimum viable population
RU	reproductive unit
SCI	Site of Community Importance
SCP	systematic conservation planning
SPA	Special Protection Area
UTM	Universal Transverse Mercator

General Introduction

1 Biodiversity and its conservation in protected areas

Biodiversity is the variability among living organisms on earth and the ecological complexes they are part of. It encompasses diversity within species, between species, and of ecosystems (Millennium Ecosystem Assessment, 2005).

The conservation of biodiversity can be motivated by a wide spectrum of values. These values range from relatively intangible ones such as aesthetic, cultural, or existence values through to more material ones such as option value – the potential for all elements of biodiversity to provide goods and services to humans – and insurance value - the role that biodiversity may play in enhancing the resilience of ecosystems in the face of global environmental change (Maclaurin and Sterelny, 2007).

Biodiversity is dramatically affected by human alterations of ecosystems (Butchart et al., 2010; Mace et al., 2005). Humans have increased species extinction rates over the past few hundred years by about 1,000 times relative to the background rates that were typical over the history of the earth (Millennium Ecosystem Assessment, 2005).

Significant political commitments for the conservation of biodiversity were made in the year 2002 with the Convention on Biological Diversity worldwide and in the year 2003 with the Kiev Resolution on Biodiversity on a Pan-European level. Targets for significantly reducing or even halting the rate of biodiversity loss by the year 2010 were agreed upon. Recent analyses show that Europe and the world have failed to meet these targets (Butchart et al., 2010; European Environment Agency, 2009; Secretariat of the Convention on Biological Diversity, 2010). Despite some local successes, the rate of biodiversity loss does not appear to be slowing (Butchart et al., 2010).

Maintaining viable populations in natural ecosystems through the creation of protected areas is widely regarded as one of the most efficient ways to protect endangered biodiversity (Bruner et al., 2001; Chape et al., 2005; Groves, 2003). The World Conservation Union defines a protected area as ‘an area of land and/or sea especially dedicated to the protection and maintenance of biological diversity, and of natural and associated cultural resources, and managed through legal or other effective means’ (IUCN, 1994). Numerous national and international regulations and laws, i.e. the Convention on Biological Diversity, the Endangered Species Act in the United States, and the Birds and Habitats Directive in the European Union, consider protected areas as being central to any conservation strategy. At present, about 12.9% of the global terrestrial area lies within formally protected areas (Jenkins and Joppa, 2009). In the European Union, about 17% of the land area is designated as protected under the Natura 2000 network (European Commission, 2009).

2 A new discipline: systematic conservation planning

Given the importance placed on protected areas, the evaluation of their effectiveness in representing and maintaining biodiversity has increasingly reached attention in the field of conservation biology. One of the consequences of this research need has been the development of a new discipline called systematic conservation planning (Margules and Pressey, 2000; Margules and Sarkar, 2007; Possingham et al., 2000). Systematic conservation planning provides tools to identify priority areas for conservation. It can be defined as a structured, target-driven approach that provides the context to account for two basic principles of any system of protected areas: (i) representativeness, the need to capture the full variety of biodiversity at all levels of organization; and (ii) persistence, the long-term survival of species and ecosystems (Margules and Pressey, 2000; Sarkar et al., 2006).

According to Margules and Pressey (2000), systematic conservation planning can be separated into six stages. The starting point is the compilation of data on the biodiversity of the planning region. Second, conservation goals for the planning region have to be identified. The next step comprises a review of the existing protected areas. In the following, additional conservation areas are selected. After implementing the conservation actions on the ground, the final step is to maintain the required values of the protected areas.

The majority of studies published in the field of systematic conservation planning concentrate on its underlying quantitative methods and approaches. The emphasis hereby lies on the reviewing of existing conservation areas and the selection of additional ones (Moilanen et al., 2009). A number of studies have shown that existing protected areas frequently do not represent the biodiversity of a region adequately (Pressey et al., 1993; Rodrigues et al., 2004; Scott et al., 2001). Their selection is often biased towards well-surveyed taxa such as birds and other vertebrate species (Hazen and Harris, 2007; Kerley et al., 2003; Polasky et al., 2001) or economically marginal landscapes (Araujo et al., 2007; Pressey, 1994; Pressey et al., 2002).

Several conservation planning software platforms are available to perform these kinds of analyses. The most widely distributed planning tool is Marxan (Ball et al., 2009). Marxan is formulated to identify sets of planning units that meet a number of representational targets at minimum cost. It was i.e. used as decision support tool for the largest successful real-world application of systematic conservation planning principles so far, the rezoning the Great Barrier Reef in Australia (Fernandes et al., 2005). Other frequently used planning tools are Zonation

(Moilanen, 2007), C-Plan (Pressey et al., 2005), and ResNet (Garson et al., 2002). All these planning tools use the principle of complementarity to ensure that the planning units prioritized for conservation actions contribute unrepresented biodiversity features to an existing set of planning units (Possingham et al., 2006). Characteristic of the mentioned software platforms is that the distinct planning units the tools are based on can only be selected in their entirety as priority area for conservation.

3 Contributions and outline of this thesis

3.1 The application of systematic conservation planning in Europe

Systematic conservation planning has not often been applied to European conservation issues (Gaston et al., 2008; Rondinini and Pressey, 2007). There are three major reasons. First, systematic conservation planning as a relatively new subdiscipline of conservation biology has evolved largely in Australia and South Africa, where human population densities are relatively low and land use patterns are often maintained for long periods (Rondinini and Pressey, 2007). In Europe, the context for planning is different. Dense human population and rapid land use change with an associated high habitat fragmentation provide comparably low opportunities for extensive conservation planning and the establishment of new or complemented reserve systems (Gaston et al., 2008; Hoekstra et al., 2005; Plieninger et al., 2006). Second, decision-making processes on conservation issues in Europe are established at continental, European Union, national, regional, and local levels. The institutional and administrative complexity complicates planning and coordination (Jongman et al., 2004; Prendergast et al., 1999). Third, although the biodiversity of the European continent is relatively well-surveyed compared to other world regions, coarse resolution data on the distribution of biodiversity hamper scientifically sound conservation planning (Araujo et al., 2005; Gaston et al., 2008). Frequently used conservation planning tools can hardly be applied straightforward in this context.

This thesis aims to facilitate and strengthen the application of systematic conservation planning methods to European conservation problems. The studies of this thesis address several of the mentioned constraints, but contribute especially to overcome the problem of planning on the basis of coarse-scale biodiversity data. The applied reserve selection model presents a methodology to calculate reserve sizes endogenously. Hereby it is possible to conduct precise spatial conservation planning despite given data deficiencies. This achievement facilitates to give advice on optimal levels of geopolitical coordination in conservation questions (see Chapter I) and enables a comprehensive assessment of the existing system of protected areas in the European Union (see Chapter IV). However, the need for better data is still emphasized in a study dealing with the benefits of high resolution global earth observation data (see Chapter III).

3.2 Incorporating economic concepts into conservation planning

In systematic conservation planning, economic factors such as conservation costs are often considered secondary to biological factors, are analyzed in post hoc assessments, or assumed to be spatially homogenous (Carwardine et al., 2008; McDonnell et al., 2002). However, the costs of conservation are, just like the distribution of biodiversity, spatially heterogeneous (Ando et al., 1998; Balmford et al., 2003). These costs may include acquisition costs, management costs, transaction costs, and opportunity costs (Naidoo and Adamowicz, 2006; Naidoo et al., 2006). Studies that consider conservation costs typically select one or several components of these costs as a surrogate measure for total costs (Adams et al., 2010). Balmford et al. (2003) estimate that acquiring land for conservation is likely to exceed subsequent costs by large factors. Conservation planning studies that explicitly incorporate land acquisition costs demonstrate considerable cost savings in meeting conservation objectives (Ando et al., 1998; Naidoo and Iwamura, 2007; Polasky et al., 2001).

This thesis introduces a method to treat land acquisition costs endogenously in reserve selection to account for market feedbacks (see Chapter II). In addition, a study presents a method to derive spatially explicit land rent data from available datasets (see Chapter III). The thesis thereby contributes to a better understanding and correct implementation of economic concepts in conservation planning applications.

3.3 The HABITAT model – conservation planning for European wetlands

Core of this thesis is the development of the HABITAT model; a reserve selection tool explicitly designed for the special requirements for conservation planning on the European continent with its fragmented habitats and high human population density. HABITAT is a mathematical programming model written in General Algebraic Modeling System (GAMS). Spatial input data are pre-processed in ArcGIS. Figure 1 gives a structural overview of the model.

The studies carried out with the HABITAT model focus on European freshwater wetland biodiversity. Wetlands were chosen as the ecosystems of focus due to two main reasons: First, freshwater wetlands are of outstanding importance for biodiversity conservation (Bobbink et al., 2006; Mitsch and Gosselink, 1993; Schweiger et al., 2002), but also play prominent roles in carbon storage (Belyea and Malmer, 2004; Zhou et al., 2007) and provision of water-related ecosystem services (Brauman et al., 2007). Despite their significance for conservation and related environmental objectives, wetlands are severely threatened by human disturbances. Over the last century, the number and size of European wetlands has decreased progressively (Jones and Hughes, 1993; Wheeler et al., 1995). Second, this thesis could build upon previous work by

Schleupner (2010). Her geographically estimated high resolution wetland habitat data on European scale serve as an important input dataset for the HABITAT model.

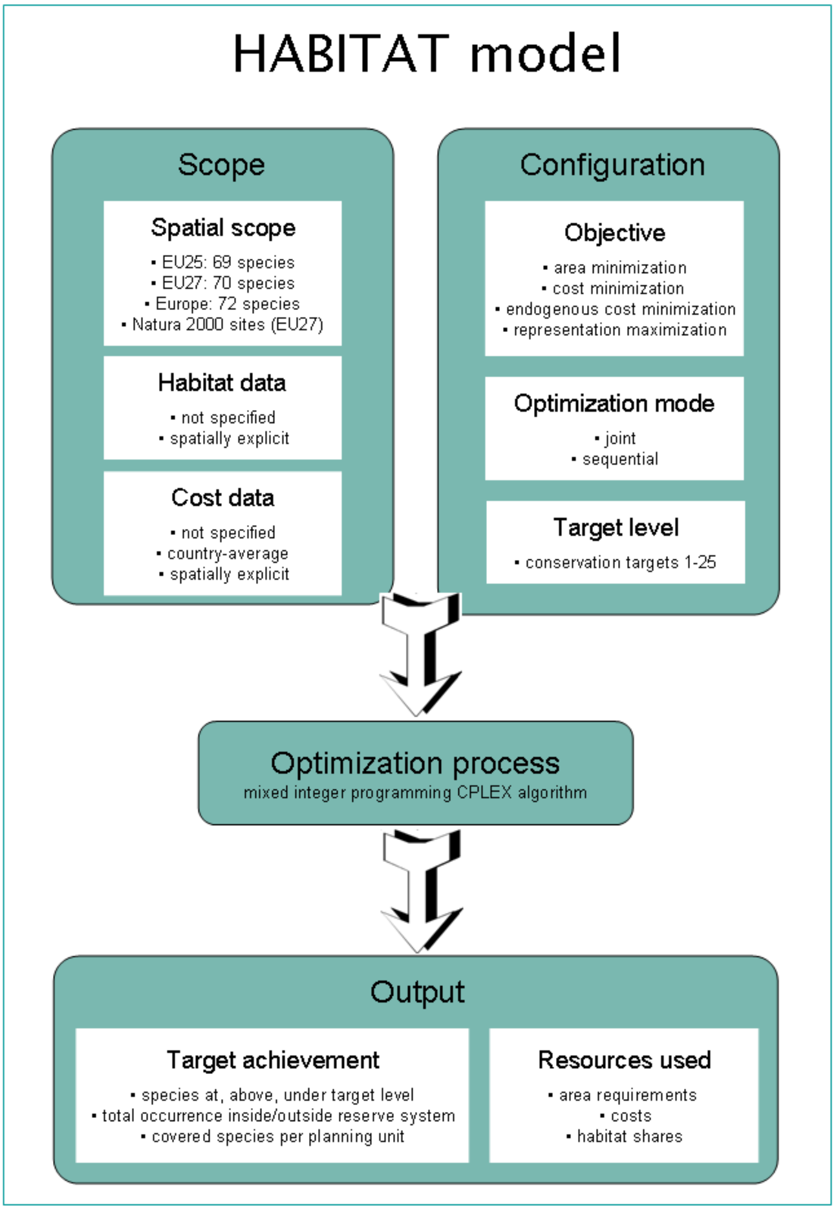

Figure 1: Overview of the HABITAT model

As most conservation planning tools, HABITAT is based on the set-covering problem. This central component of the systematic conservation planning philosophy aims at efficiency of resource use (Margules and Pressey, 2000). The set-covering problem and its derivatives have been studied in the fields of operations research and location science since the 1970s (Marianov et al., 2008). Margules et al. (1988) reformulated it in the context of conservation planning. The objective is to find a set of conservation sites that achieves a conservation target at minimum cost.

The basic formulation of this problem is

$$\min \sum_{i=1}^{N} c_i x_i$$

$$\text{s.t.} \quad \sum_{i=1}^{N} x_i a_{ij} \geq r_j \ , \qquad \text{for all features } j,$$

where a_{ij} is the occurrence level of feature j in site i, c_i is the cost of site i, N is the total number of sites, and r_j is the representation level for feature j. The binary variable x_i has a value of 1 for sites included in the selection and 0 otherwise.

3.4 Overview of studies and chapters of this thesis

The thesis' chapters are based on four research papers. Each paper was submitted to international peer-reviewed journals as well as presented at international meetings and conferences.

I Jantke, K. and U.A. Schneider (2010), *Multiple-species conservation planning for European wetlands with different degrees of coordination*, published in *Biological Conservation*, 143 (7), pp. 1812-1821.

This paper was presented at the EURECO – GFOE in Leipzig, Germany (September 2008) and the Annual Retreat of the International Max Planck Research School on Earth System Modelling in Lüneburg, Germany (September 2008). The paper investigates different degrees of geopolitical coordination in multiple-species conservation planning. Reserve sizes are represented endogenously in the optimization model. The analysis illustrates and quantifies the efficiency of multi-species conservation activities.

II Jantke, K. and U.A. Schneider (2010), *Integrating land market feedbacks into conservation planning – a mathematical programming approach*, under review in *Environmental Modeling & Assessment*.

This paper was presented at 22nd Annual Meeting of the Society for Conservation Biology in Chattanooga (Tennessee), USA (July 2008) and is accepted for presentation at the 11th Biennial Conference of the International Society for Ecological Economics in Oldenburg and Bremen, Germany (August 2010). The study demonstrates a method to integrate land market feedbacks directly and consistently into conservation planning tools. To illustrate the effect of incorporating the dynamic nature of opportunity costs, different cost representations are compared in a multiple-species conservation planning exercise.

III Jantke, K., C. Schleupner, and U.A. Schneider (2010). *Benefits of global earth observation for conservation planning in the case of European wetland biodiversity,* submitted to *Environmental Conservation.*

This paper was presented at the GEO-BENE Project Meeting in Laxenburg, Austria (June 2008) and at the 33rd International Symposium on Remote Sensing of Environment in Stresa, Italy (May 2009). The study investigates different degrees of errors related to the employment of coarse scale land cover and land value information in conservation planning. It contributes to the benefit assessment of global earth observation in the realm of biodiversity and ecosystems.

IV Jantke, K., C. Schleupner, and U.A. Schneider (2010). *Gap analysis of European wetland species: priority regions for expanding the Natura 2000 network,* submitted to *Biodiversity and Conservation.*

This paper was presented at the 2nd Evaluation of the International Max Planck Research School on Earth System Modelling in Hamburg, Germany (April 2010) and is accepted for presentation at the 24th International Congress for Conservation Biology in Edmonton (Alberta), Canada (July 2010). The paper evaluates the performance of the existing Natura 2000 system in covering endangered wetland vertebrate species. It identifies potentials for expanding the network to move toward complete coverage and presents spatially explicit priority regions for a cost-effective expansion.

The style of chapters I-IV is kept according to the submitted or published manuscripts, following the selected journal style. The corresponding references and appendices are presented at the end of each chapter. The literature sources of the ecological model input data for all 72 wetland vertebrate species and their vernacular names are presented in separate appendices at the end of the thesis.

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I

Multiple-species conservation planning for European wetlands with different degrees of coordination

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Abstract. Selection and establishment of reserves was often done unplanned and uncoordinated between regions. Systematic conservation planning provides tools to identify optimally located priority areas for conservation. Planning for multiple species promises adequate provision for the needs of a range of threatened species simultaneously. Several studies apply the set-covering problem by minimizing resources for given conservation targets of multiple species. We extend this method by also considering different degrees of coordination in multiple-species conservation planning and representing reserve sizes endogenously. A deterministic, spatially explicit programming model solved with mixed integer programming is used to represent minimum habitat area thresholds for all included biodiversity features. The empirical model application to European wetland species addresses five different scenarios of coordination in conservation planning, including taxonomic, political, and biogeographical coordination of planning. Our approach illustrates and quantifies the efficiency of multi-species conservation activities. We show that maximum coordination in conservation planning enhances area efficiency by 30% compared to no coordination. Furthermore, strong coordination in conservation planning does not only reduce the area requirement, but synergy effects even enable the conservation features to achieve higher conservation objectives. Spatial subdivision of planning, however, leads to highest area requirements and less conservation target achievement.

Keywords: systematic conservation planning, set-covering problem, representation, persistence, mixed integer programming, European environmental policies

1 Introduction

Protected areas are often established ad hoc without coordination between regions (Gonzales et al., 2003; Margules and Pressey, 2000; Pressey, 1994). In the European Union, the Natura 2000 network of protected areas currently covers about 17% of the total land area (European Commission, 2009). Hoekstra et al. (2005) identify the vast majority of the European continent's terrestrial area as crisis ecoregions with extensive habitat degradation and limited habitat protection. National governments in the European Union and the European Commission apply different strategies of conservation planning. There are protection plans for selected single species (Amstislavsky et al., 2008; Koffijberg and Schaffer, 2006; Tucakov et al., 2006), species groups (Goverse et al., 2006; Lovari, 2004; Papazoglou et al., 2004) as well as national conservation programs (Elliott and Udovc, 2005; Sepp et al., 1999; Vuorisalo and Laihonen, 2000). Transfrontier national parks covering characteristics of specific biogeographical regions are located for instance in mountainous regions (Oszlanyi et al., 2004; Williams et al., 2005b). Important pan-European initiatives (see Jones-Walters (2007) for a review on European ecological networks) are the Natura 2000 network based on the Birds and Habitats Directives (79/409/EEC; 92/43/EEC) and the Emerald's network based on the Bern convention (Council of Europe, 1979).

In light of increasing opportunity costs for land, questions on the efficiency of existing conservation strategies arise. The main question we address in this study is: How efficient in terms of area requirement are different strategies of coordination in conservation planning?

Systematic conservation planning (SCP) provides tools to identify priority areas for conservation (Margules and Pressey, 2000; Margules and Sarkar, 2007; Possingham et al., 2000). Formulated as minimization problem, SCP optimizes the allocation of conservation areas such that the total requirement of resources (typically, area or costs) under a given conservation target is minimized (McDonnell et al., 2002; Possingham et al., 2000; Williams et al., 2005a). Previous studies estimate the optimal arrangement of protected areas for exogenously given conservation targets (ReVelle et al., 2002; Saetersdal et al., 1993; Tognelli et al., 2008).

Several studies point out that the focus in reserve site selection lies on representation of biodiversity features whereas persistence is often inadequately addressed (Cabeza and Moilanen, 2001; Haight and Travis, 2008; Önal and Briers, 2005; Williams et al., 2005a). We extend the set-covering problem by: (i) combining representation and persistence requirements and (ii) representing the reserve sizes endogenously. As proposed by Marianov et al. (2008), we thereby account for species-specific habitat area needs to enable viable populations.

Whether setting definitive and measurable conservation targets is possible and reasonable has been discussed controversially (Soulé and Sanjayan, 1998; Tear et al., 2005; Wilhere, 2008). We do not determine a single representation target as sufficient for the long-term protection of the considered biodiversity features, but rather estimate a relationship between a relatively wide range of representation targets and their overall area requirement. There are three major reasons. First, we cannot endogenously determine the optimal conservation target because we do not estimate the benefits of conservation. Second, alternative target levels provide additional insight, which may help researchers and policymakers in finding the preferred conservation targets. Third, the costs of simulating additional targets are low and involve mainly computational costs. Justus et al. (2008) adopt a similar approach for representing biodiversity surrogates in five regions.

Multiple-species conservation planning has been discussed elaborately elsewhere (McCarthy et al., 2006; Moilanen et al., 2005; Nicholson and Possingham, 2006). However, most previous studies have neither explicitly examined different degrees of multiple-species conservation planning nor quantified the area reduction potential resulting from comprehensive coordination. First insights into efficiency gains from coordination in Europe give Strange et al. (2006) and Bladt et al. (2009). For North America, first studies on the impact of different spatial extents in planning provide Vazquez et al. (2008) and Pearce et al. (2008).

A deterministic, spatially explicit programming model solved with mixed integer programming is used to represent minimum habitat area thresholds for all included biodiversity features. Whether to prefer iterative heuristics or exact algorithms in reserve selection has been covered extensively (Pressey et al., 1996; Rosing et al., 2002; Vanderkam et al., 2007). In contrast to alternative methods, the chosen mixed integer programming with its branch-and-bound algorithm reveals at any time the quality of the solution with respect to a best possible integer solution. Our model quantifies area requirements for conservation under different assumptions of coordinated planning. We apply scenarios which mimic commonly used conservation strategies in Europe and globally. The analysis is done for European wetland species but is easily adaptable to other species, biodiversity features, or regions.

2 Methods

2.1 Integrating representation and persistence: the conservation target

Successful conservation requires consideration of both representation and persistence (Margules and Pressey, 2000; Sarkar et al., 2006). Each species in our model has to achieve exogenously assigned representation targets which can differ across species. The persistence criterion is subject to two conditions. First, each species' representation corresponds to one minimum viable population (MVP). A population is considered viable when the allocated land area equals smallest the minimum critical area (MCA) which is defined as follows:

MCA = density * MVP size for all species.

The species-specific measure of MCA depends on density data and proxies for MVP sizes. Density data can differ substantially depending on habitat quality (Foppen et al., 2000; Riley, 2002) or due to bias in sampling effort (Schwanghart et al., 2008). To account for that variability, we solve the model for different density data. We assume that species do not affect each others densities. Also, we do not explicitly portray competition between species. The second persistence condition requires that the land area that corresponds to a species' MCA is allocated to appropriate habitat types. We therefore classify the included habitat types species-specific as either necessary for its survival, as optional habitats, or as unsuitable.

2.2 Planning units

Our model is spatially explicit with planning units differing in shape and size. There are two possible states of each planning unit; it is either used as a species' reserve (1) or not (0). Status (1) is only achievable if a species was historically observed in a planning unit. The potential reserve areas are determined for each planning unit. However, using a planning unit for conservation does not necessarily allocate the entire planning unit's reserve area. Only those fractions of planning units are selected which are necessary to fulfill the respective conservation target. On the other hand, the potential reserve area within a single planning unit may not be sufficient for wide-ranging species. These species are therefore allowed to inhabit further habitat in adjacent planning units. This procedure allows easy implementation of planning units with varying sizes. Persistence criterions can be addressed regardless of the planning unit's size. We assume constant habitat suitability across all possible planning units.

2.3 Mathematical optimization model

The formal framework follows and expands the set-covering problem. We use the following notation: $p = \{1,\dots,P\}$ is the set of planning units; $t = \{1,\dots,T\}$ is the set of habitat types; $q = \{1,\dots,Q\}$ is the set of different habitat qualities; and $s = \{1,\dots,S\}$ is the set of species. In addition we employ several set mappings, which contain possible combinations between two or more indexes. In particular, $u(t,s)$ identifies the mapping between species and required or optional habitat types and $k(p,t,s)$ possible existence of species and habitats in each planning unit. The objective variable Z represents the total habitat area in hectares. The decision variable $Y_{p,t,q}$ determines the habitat area per planning unit p, habitat type t, and habitat quality q in hectares. $X_{p,s}$ is a binary variable with $X_{p,s} = 1$ indicating species s is protected in planning unit p, and $X_{p,s} = 0$ otherwise. $a_{p,t,q}$ is the maximum available area to be selected per planning unit p, habitat type t and habitat quality q. $d_{q,s}$ represents species- and habitat quality-specific density data. m_s is a species-specific proxy for MVP size. $h_{t,s}$ determines which habitat types t are required by species s. r_s is the representation target per species s. v_s specifies deviations from the representation target based on exogenous maximum occurrence calculations.

Minimize $$Z = \sum_{p,t,q} Y_{p,t,q} \qquad (1)$$

subject to:

$$Y_{p,t,q} \leq a_{p,t,q} \qquad \text{for all p,t,q} \qquad (2)$$

$$\sum_{t,q} d_{q,s} \cdot Y_{p,t,q}\Big|_{k(p,t,s)\wedge u(t,s)} \geq m_s \cdot X_{p,s} \qquad \text{for all p,s} \qquad (3)$$

$$\sum_{q} Y_{p,t,q} \geq h_{t,s} \cdot X_{p,s} \qquad \text{for all p,t,s} \qquad (4)$$

$$\sum_{p} X_{p,s} \geq r_s - v_s \qquad \text{for all s} \qquad (5)$$

$$\sum_{p,t,q} d_{q,s} \cdot Y_{p,t,q}\Big|_{k(p,t,s)} \geq r_s \cdot m_s \qquad \text{for all s.} \qquad (6)$$

The objective function (1) minimizes the total habitat area across planning units, habitat types, and site qualities. Constraint (2) limits habitat areas in each planning unit to given endowments. Constraint (3) ensures that the habitat area for the conservation of a particular species is large enough to support viable populations of that species. The constraint portrays minimum area requirements for all protected species in all planning units. The summation over habitat types depicts the choice between possible habitat alternatives. Constraint (4) forces the existence of required habitat types for all species which are chosen in a particular planning unit. Constraint (5) implements the representation targets for all species. This constraint allows deviations from the target if the number of planning units with occurrence data is below the representation target. Constraint (6) ensures that the total population size equals at least the representation target times the MVP size. This constraint is especially relevant for cases where the representation target is higher than the number of available planning units for conservation. For example, a representation target of ten viable populations with possible species occurrences in only nine planning units would under (6) require at least one planning unit to establish enough habitat for two viable populations.

The problem is solved with mixed integer programming using the General Algebraic Modeling System (GAMS) software version 22.9.

3 Biodiversity conservation on European wetlands

Freshwater wetlands are of outstanding importance for biodiversity conservation (Bobbink et al., 2006; Mitsch and Gosselink, 1993; Schweiger et al., 2002). They also play prominent roles in carbon storage (Belyea and Malmer, 2004; Zhou et al., 2007) and provision of water-related ecosystem services (Brauman et al., 2007). However, wetlands are severely threatened by human disturbances (Bobbink et al., 2006; Bronmark and Hansson, 2002). Recognizing their significance for conservation and related environmental objectives, we apply our model to freshwater wetlands.

3.1 Data

Freshwater wetland dependent species serve as surrogates for biodiversity. We consider 70 tetrapod wetland species which appear in the appendices of the Birds and the Habitats Directive (79/409/EEC; 92/43/EEC). The species assemblage includes 16 amphibian, 4 reptile, 41 breeding bird, and 9 mammal species. Recorded occurrences identify their European distribution. These data originate from the Atlas of Amphibians and Reptiles in Europe (Gasc et al., 1997), the EBCC Atlas of European Breeding Birds (Hagemeijer and Blair, 1997), and the Atlas of European Mammals (Mitchell-Jones et al., 1999).

Density data for all 70 species are equal to the maximum observed densities from a comprehensive literature review. In addition, we use the proposed standards for minimum population sizes from Verboom et al. (2001) as proxies for MVP size. These population sizes depend on species' body sizes and life expectancy. One MVP in our model represents 120 reproductive units of long lived or large vertebrates and 200 reproductive units of other vertebrates. Reproductive units correspond to pairs, territories, or families of a species.

Five broad wetland habitat types appear in our dataset, namely mire, wet forest, wet grassland, water course, and water body. "Open water" as a sixth type is assigned to species that either require water courses or water bodies. Information on species' habitat type requirements are also taken from the literature. We distinguish required and optional habitat types. See Appendix A for the ecological data of the 70 wetland species

The dataset covers 25 out of 27 European Union member states (see Figure I-1). Cyprus is excluded from the analysis due to the lack of comprehensive atlas data of all species; Malta is eliminated as none of the considered species have records in the used data sources. Furthermore, the Macaronesian islands are excluded due to general lack of data.

The resolution of the planning units is consistent with that of the species occurrence data. The Universal Transverse Mercator (UTM) projection results in grid squares of about 50 km edge length. The terrestrial parts of all 2235 grid cells belonging to the selected European countries serve as planning units. In this model version we allow the allocation of the entire unsealed land area in each planning unit to the five relevant habitat types. As we restrict habitat establishment to those planning units where a species was observed historically, we implicitly integrate the necessary natural conditions for the existence of these habitats.

Figure I-1: Spatial scope of empirical model application. The scope includes 25 of 27 European Union member states. Malta, Cyprus, and Macaronesia are excluded due to lack of biodiversity data.

3.2 Conservation planning scenarios

We define coordination of conservation planning as solving the set-covering problem simultaneously for different species. Five broad categories of coordination are distinguished which contain one or more independent planning entities. Within each entity, the model minimizes the habitat area requirements of all associated species jointly.

There are two reference scenarios delineating the lower and upper boundaries of possible solutions for scenarios without spatial segregation of planning within the European Union. The most uncoordinated scenario assumes that preservation of each of the considered 70 species is planned independently. Hereafter, we refer to this scenario as **no coordination in conservation planning**. The other extreme scenario involves the case of **maximum coordination in conservation planning**. This ideal scenario represents the maximum possible simultaneous conservation for our model. We assume completely coordinated planning for all included wetland species.

Furthermore, we analyze three intermediate scenarios to simulate the impacts of political, biogeographical, and taxonomic coordination limits. Political and biogeographical region based coordination implies dividing the European Union into sub-units. In the first scenario, we apply **coordinated conservation planning within countries**. Each European member state jointly protects all species which occur in large parts on its territory (see Appendix B); thus we have a political division of planning. However, there is no coordination between countries. The second intermediate scenario examines **coordinated conservation planning within biogeographical regions**. There are seven biogeographical units in the European Union which include the Alpine, Atlantic, Black Sea, Boreal, Continental, Mediterranean, and Pannonian region (see Appendix C). Finally, the third intermediate scenario coordinates conservation planning only within tetrapod classes. We consider the four taxon groups amphibians, reptiles, birds, and mammals as entities for each of which independent plans are developed. Hereafter, we refer to this scenario as **coordinated conservation planning within taxonomic groups**. For no coordination across species or coordination within taxonomic groups, we use a special algorithm to make the individually obtained solutions compatible. In particular, we first determine the order in which protection plans for the different species or taxonomic groups are established. To guarantee that the individual solutions can be combined without violating land endowments, we require that the allocated habitat areas under each established protection plan remain fixed for all subsequent plans.

4 Results

The habitat allocation model minimizes the total area of protected habitats for different conservation targets. Figure I-2 shows the total wetland area requirements and the optimal allocation of reserves to alternative wetland types under maximum coordination in conservation planning across 70 species. The area is shown in million hectares for conservation targets ranging between 1 and 20 population representations. The optimal share of habitat types varies between different targets. The highest amount of land is allocated to water bodies and wet grasslands for most displayed targets.

Figure I-3 compares the total area requirements of all five conservation planning scenarios. Spatial subdivision of the planning scope into countries or biogeographical regions implies highest area requirements. However, coordinated planning within biogeographical regions falls behind for the upper displayed targets. Maximum coordination results in the lowest area requirement throughout the targets. This scenario saves on average 42 percent relative to the most area-intensive one and 25 percent relative to the scenario without any coordination. Note that we show mean values from five model runs for the scenario without coordination.

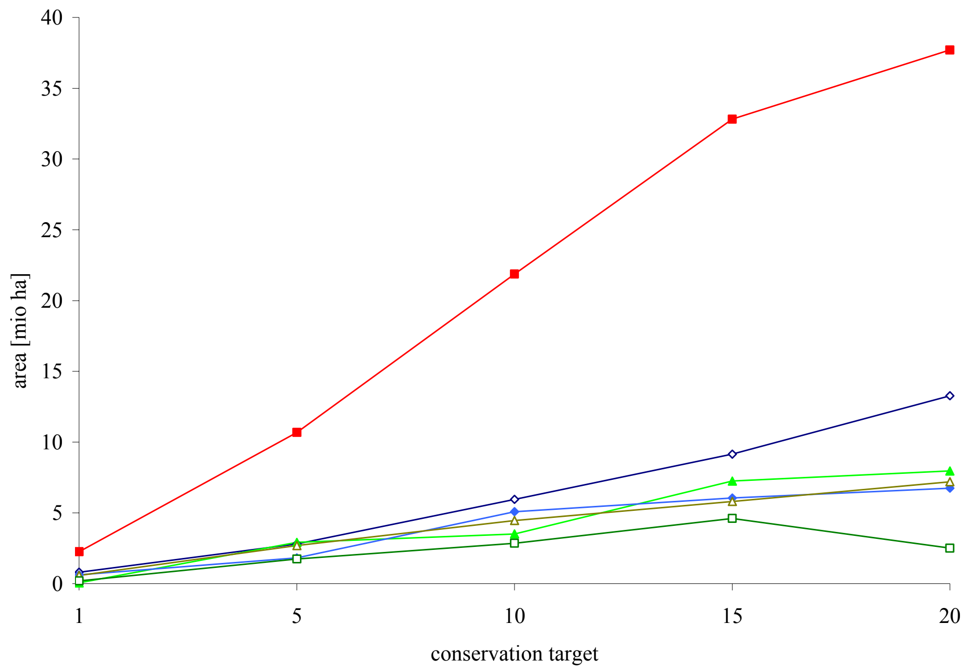


Figure I-2: Maximum coordination in conservation planning: allocation to wetland habitat types and total area requirement. The upper curve shows the minimum total area in million hectares needed to ensure the conservation targets 1 to 20. The lower curves display the shares of the five included wetland habitat types which add up to the total required area. The habitat types comprise water bodies (open diamonds), wet grasslands (solid triangles), water courses (solid diamonds), mires (open triangles), and wet forests (open squares).

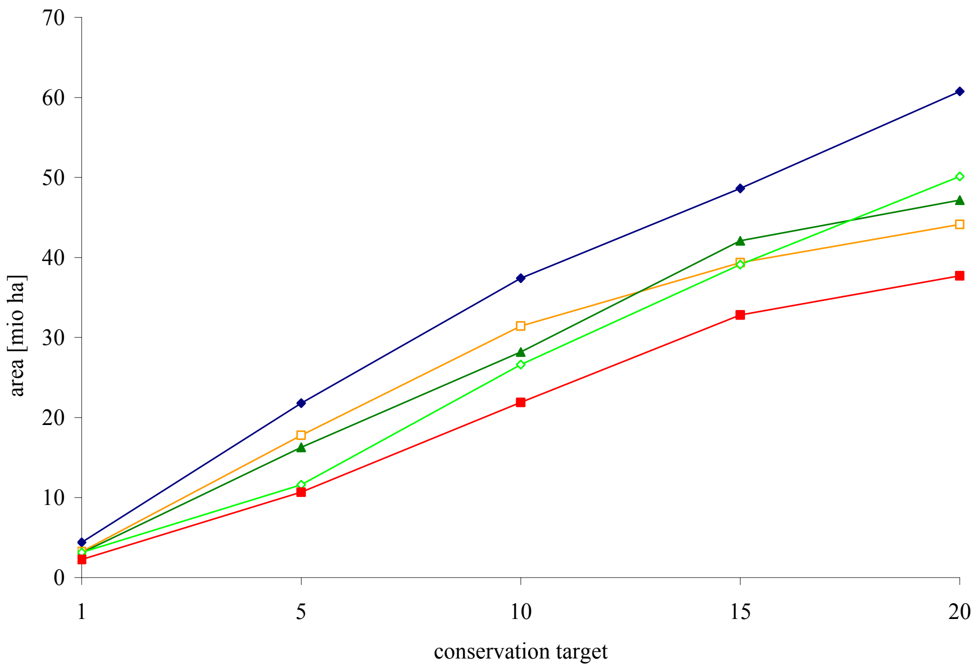


Figure I-3: Total area requirements for five scenarios of coordinated conservation planning. Shown is the wetland reserve area in million hectares that is required to represent 1 to 20 viable populations of the 70 included species in Europe. The curves indicate area requirements for the five scenarios of coordinated planning within countries (solid diamonds), coordinated planning within biogeographical regions (open squares), no coordination (solid triangles), coordinated planning within taxonomic groups (open diamonds), and maximum coordination (solid squares).

Table I-1 displays major results for the conservation targets 1, 10, and 20. The three spatially all-embracing conservation strategies – no coordination, maximum coordination, and coordination within taxonomic groups – are more area-efficient as they cover more species on less habitat area throughout the targets. However, the scenarios no coordination and coordination within taxonomic groups still fall far short behind the performance of the maximum coordination scenario.

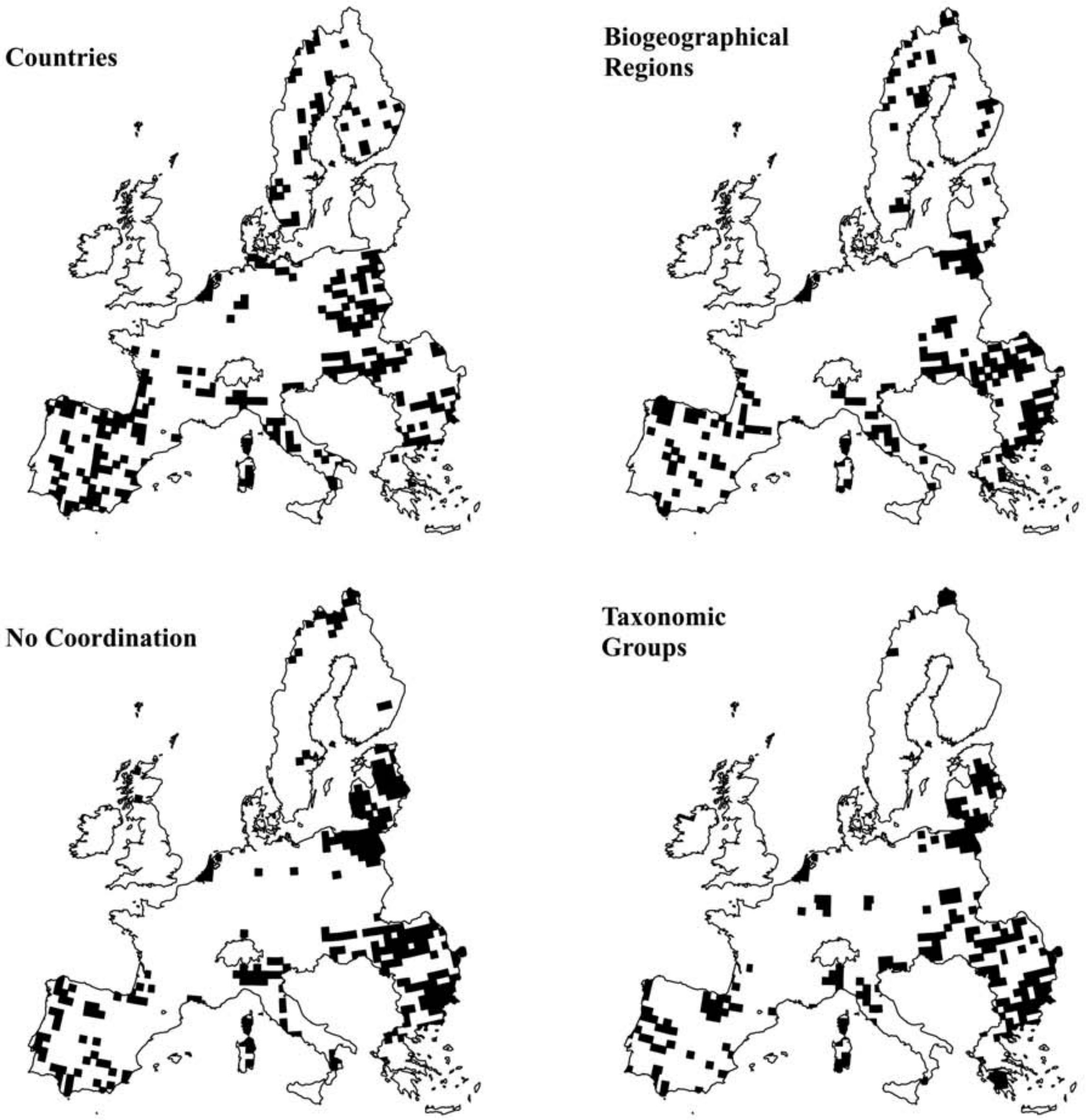
Table I-1: Key results of scenarios of coordination. Shown are the numbers of planning units (P=2235) in which reserve area is allocated, average and maximum numbers of species that these planning units contain, as well as the total allocated area for selected conservation targets.

Scope of coordination in conservation planning		Countries			Biogeographical Regions			No Coordination			Taxonomic Groups			Maximum Coordination		
Conservation target		1	10	20	1	10	20	1	10	20	1	10	20	1	10	20
Selected planning units		46	329	602	40	292	504	86	335	524	55	292	499	25	215	346
Covered species per planning unit	Average	1.8	2.2	2.2	2.5	2.5	2.6	3.6	3.5	3.5	2.4	3.5	3.6	3.8	4.0	4.4
	Maximum	9	10	11	12	13	20	18	21	23	14	20	21	23	23	23
Total area (mio ha)		4.4	37.4	60.8	3.2	31.4	44.1	3.1	28.2	47.2	3.1	26.6	50.1	2.3	21.9	37.7

Target achievement differs substantially between the scenarios with and without spatial segregation (Table I-2). The higher the conservation target in the country- and biogeographical region-scenario, the fewer species are able to fulfill it. Also, only few species exceed the target. Target achievement does not differ remarkably between the three spatially all-embracing scenarios. The majority of species is represented according to the respective target. About one third of all species exceeds the conservation target by a factor of 2 or higher. Figure I-4 shows the spatial distribution of selected planning units for all five scenarios exemplarily for conservation target 10.

Table I-2: Performance of scenarios in conservation target achievement. Shown are the numbers of species (S=70) that underachieve, fulfill, or exceed selected conservation targets. Note that target underachievements occur when species' area requirements for viable populations cannot be fulfilled according to the required target.

Scope of coordination in conservation planning		Countries			Biogeographical Regions			No Coordination			Taxonomic Groups			Maximum Coordination		
Conservation target		1	10	20	1	10	20	1	10	20	1	10	20	1	10	20
Number of species below, at, and above the specified target	< 100%		4	10		3	9									
	100%	**63**	**48**	**41**	**53**	**51**	**47**	**46**	**45**	**49**	**50**	**41**	**47**	**54**	**50**	**47**
	≤ 200%	4	15	17	11	14	13	5	11	13	8	20	18	10	16	19
	≤ 300%	2	3	2	3	1	1	4	5	4	6	5	1	3	2	2
	> 300%	1			3	1		15	9	4	6	4	4	2	2	2



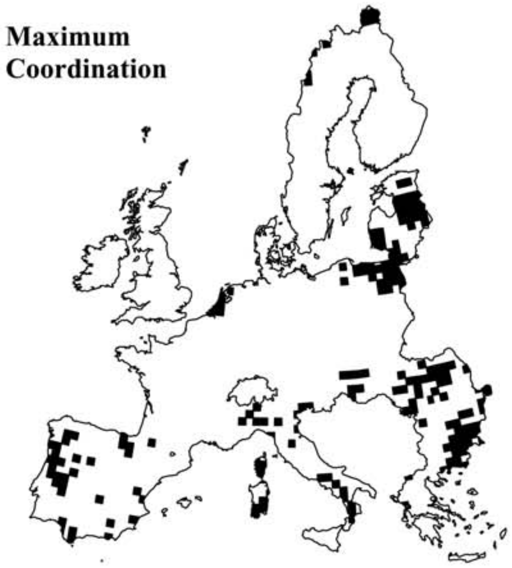


Figure I-4: Spatial distribution of selected planning units for conservation target 10. Shown are the planning units in which habitat area is to be established under the different assumptions of coordinated planning. The allocated habitat area per planning unit ranges between 10 ha and 250.000 ha.

5 Discussions and conclusions

5.1 *Efficiency of multiple-species conservation planning: conservation implications*

Our analysis measures the area efficiency of simultaneous conservation planning. The magnitude of our results confirms that conservation planning should be coordinated at the largest possible spatial scale of an ecozone. These findings are in accordance with results of a study by Bladt et al. (2009) analyzing coordinated conservation efforts at a European scale. However, full coordination requires a high degree of collaboration between many organizations from different countries and incurs transaction costs. These costs are not included in our study. On the other hand, there may be economies of scale by avoiding the parallel development and maintenance of a large number of protection plans. Our simulations show that even small degrees of coordination can lead to substantial synergies. A study by Strange et al. (2006), comparing national and regional conservation strategies in Denmark, supports this conclusion also with respect to cost-efficiency. An additional advantage of joint conservation effort is according to our study that a lot of species are represented several times more than the respective conservation target enforces. The spatial scope of coordinated planning can greatly affect location and size of priority areas for conservation. These results agree with findings by Vazquez et al. (2008) who did a similar analysis in North America. EU-national coordination does not only yield the fewest target achievements but also requires the largest habitat area for the majority of conservation targets. Note that this argument is not to question the national responsibilities for species protection (see Schmeller et al. (2008) for a review). Still, according to our results, options for cooperation beyond the borders of countries should be exploited whenever these countries belong to the same ecozone. The same argumentation holds for conservation planning across different biogeographical regions.

The relatively simple case study quantifies possible advantages from multiple-species conservation planning in terms of area requirements. Reality in conservation planning is undoubtedly more complex. An application of SCP for actual policymaking requires great care in ensuring adequate representation of each species' specific needs. Constraints should be added to the SCP model to keep valuable existing reserves in place. Encouraging to imprudently lump together basically different species is not purpose of this study as such proceeding will most likely not favor conservation success. Also, in certain circumstances coordination in planning may not be the best option for action. In particular cases, e.g. when species are directly faced with extinction, urgent action is indispensable. Hence, time lags until reserve establishment associated with comprehensive planning would discourage coordination efforts in such cases.

Highly coordinated conservation planning seems particular suitable in cases where on large spatial extent new reserve systems are to be established or current systems are to be enlarged.

5.2 Limitations in conservation planning

Applying SCP usually involves several simplifications. First, species are taken as surrogates for biodiversity. This may lead to non-optimal conservation decisions (Margules and Pressey, 2000; Rodrigues and Brooks, 2007). The necessity to include occurrence data as basic input parameter into reserve selection models leads to bias towards well-surveyed species such as vertebrates (Hazen and Harris, 2007; Kerley et al., 2003; Polasky et al., 2001). We furthermore assume that species do not influence each others' population densities, and treat each colonized planning unit as equally appropriate for a species.

In addition to these general shortcomings, two further simplifications were made. First, we do not directly account for spatial reserve design criterions such as connectivity or compactness in our model. This is especially critical for species with low dispersal abilities such as amphibians and reptiles. Polasky et al. (2008) show an approach to explicitly consider species-specific dispersal abilities within reserve selection models. Even so, simultaneous planning implicitly results in compact reserves. Also, the spatial configuration of potential reserves is of utmost importance mainly in the concrete delineation of habitats to be protected. Accurate matching of species and reserves (Araujo, 2004; Araujo et al., 2005) has to be considered carefully when downscaling our results, e.g. to propose an improved network of reserve areas. Ongoing work, however, is addressing this issue so that it may be possible to be more inclusive in the future (e.g., Schleupner and Schneider, 2008).

Second, we do not include existing wetland habitats but rather allowed the model to allocate the entire land area to the wetland habitat types which can lead to unrealistic high wetland fractions. However, preliminary simulations with geographically estimated wetland data (Schleupner, 2007) indicate that the range of results does not change markedly.

The importance of area-efficient conservation increases with the value of land. These values are heterogeneous and minimization of area requirements for reservation does not guarantee minimization of costs (Balmford et al., 2000; Naidoo et al., 2006; Polasky et al., 2001). Costs are an important factor not directly accounted for in our study as we minimize the overall habitat area instead of the land costs. However, when appropriate data on land costs are not available, conservation planning studies often use area as a substitute for costs (McDonnell et al., 2002).

5.3 *Conservation target: integrating representation and persistence*

Each conservation target does not just correspond to the presence of a biodiversity feature, i.e. a certain species, but rather to the establishment of a viable population. Unlike commonly done in conservation planning (Tognelli et al., 2008; Williams et al., 2005a; Williams and Araujo, 2002), our approach does not necessarily select the entire planning unit as a priority area for conservation. Instead, the identified habitat areas must meet the MCAs for all preserved species in each planning unit. Thus, their sum may fall short of a planning unit's total area. If the area needed for the establishment of a viable population for a wide-ranging species cannot be provided by a single planning unit, areas from adjacent planning units are added.

The above-described procedure seems reasonable when dealing with large planning units for which it is unlikely or impossible to reserve them entirely. This study, for example, uses relatively coarse species occurrence data which result in planning units with an edge length of about 50 km.

We make several simplifications to adopt the conservation target approach. First, although the model structure allows the determination of species-specific representation targets, we employ the same target for each species in our application. Main reason for it is the difficulty in consistently determining explicit targets for individual species (Kerley et al., 2003). Second, to account for persistence, reliable density data as well as proxies for MVP sizes are essential. However, density data from literature vary substantially or are biased towards regions with high population densities (Schwanghart et al., 2008). Whether using absolute numbers for viable population sizes seems appropriate is subject to further discussion (Nicholson et al., 2006; Traill et al., 2007). Note that we do not assume the utilized figures to represent real MVPs nor that defining explicit sizes for persistent populations is possible. This is particularly true for such a range of species with divergent habitat requirements. Given the lack of better data, we still use these figures as working targets in our conservation planning exercise. Similar proceeding can be found in Kautz and Cox (2001), Verboom et al. (2001), and Kerley et al. (2003). Polasky et al. (2008) use population viability thresholds to estimate the number of species sustained on a landscape. Note that we do not account for spatio-temporal aspects of persistence.

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Appendix A

Table I-A1: Wetland species of European conservation concern

Shown are the 70 included species with their proxies for MVP sizes (adapted from Verboom et al. (2001)), density data, and habitat types. The genus *Discoglossus galganoi* includes *Discoglossus jeanneae*. For *Castor fiber*, the Estonian, Latvian, Lithuanian, Finnish and Swedish populations are excluded (according to 92/43/EEC). Regarding the densities for colonial birds, we differentiate nesting and foraging areas. The foraging area is set to 5 ha per reproductive unit (RU). Regarding the densities of the amphibian species, we assume 10 RU per hectare for solitary species and 20 RU per hectare for gregarious species. x stands for a required habitat type; / stands for an optional habitat type. The category open water is introduced for species that need some type of open water habitat. Wide-ranging species are indicated with an asterisk.

Scientific name	MVP (RU)	Maximum density (RU/ha)	Required (x) and optional (/) habitat types					
			Mire	Wet forest	Wet grassland	Water course	Water body	Open water
Amphibians								
Alytes muletensis	200	20				x		
Bombina bombina	200	20			x		x	
Bombina variegata	200	20		/	/		x	
Chioglossa lusitanica	200	10				x		
Discoglossus galganoi	200	10					x	
Discoglossus montalentii	200	10				x		
Discoglossus sardus	200	10					x	
Pelobates fuscus insubricus	200	10					x	
Rana latastei	200	20		x			x	
Salamandrina terdigitata	200	10				x		
Triturus carnifex	200	10		/	/		x	
Triturus cristatus	200	10		/	/		x	
Triturus dobrogicus	200	10			/		x	
Triturus karelini	200	10					x	
Triturus montandoni	200	10		x	/		x	
Triturus vulgaris ampelensis	200	20		/	/		x	
Reptiles								
Elaphe quatuorlineata	120	2			/			
Emys orbicularis	120	15					x	
Mauremys caspica	120	9						x
Mauremys leprosa	120	9						x
Birds								
Acrocephalus paludicola	200	1.09			x			
Alcedo atthis	200	0.15						x
Anser erythropus	200	0.127		x				x
*Aquila chrysaetos**	120	0.0002	/		/			
*Aquila clanga**	120	0.000055	/	x	/	/	/	

Scientific name	MVP (RU)	Maximum density (RU/ha)	Required (x) and optional (/) habitat types					
			Mire	Wet forest	Wet grassland	Water course	Water body	Open water
Ardea purpurea purpurea	120	0.19			x			x
Ardeola ralloides	200	0.19			x		x	
Asio flammeus	200	0.1	/		/			
Aythya nyroca	200	1			x		x	
Botaurus stellaris stellaris	200	0.5			x			
Chlidonias hybridus	200	0.19			/		x	
Chlidonias niger	200	0.19			x		x	
*Ciconia ciconia**	120	0.001415			x			*x*
*Ciconia nigra**	120	0.00018		x				x
Crex crex	200	0.19	/		x	/		
Fulica cristata	200	10			x		x	
Gavia arctica	120	0.006					x	
Gelochelidon nilotica	200	0.19			x	x		
Glareola pratincola	200	8			x		x	
*Grus grus**	120	0.00043	/	/	/		/	
Haliaeetus albicilla	120	0.01273		x				x
Hoplopterus spinosus	200	0.3846			x			x
Ixobrychus minutus minutus	200	1.97			x			x
Marmaronetta angustirostris	200	0.19			x		x	
Milvus migrans	120	1.2733						x
Nycticorax nycticorax	200	0.19			x			x
Oxyura leucocephala	200	1.5					x	
*Pandion haliaetus**	120	0.0004		/			x	
Pelecanus crispus	120	0.19			/		x	
Pelecanus onocrotalus	120	0.19			/		x	
Phalacrocorax pygmaeus	200	0.19		/	/		x	
Philomachus pugnax	200	1	/		/			
Platalea leucorodia	120	0.19		/	x		x	
Plegadis falcinellus	200	0.19		/	x		x	
Porphyrio porphyrio	200	3.3			x		x	
Porzana parva parva	200	5			x		/	
Porzana porzana	200	0.333	/		/			
Porzana pusilla	200	3.5368			x			
Sterna albifrons	200	0.19				x	/	
Tadorna ferruginea	120	10					x	
Tringa glareola	200	0.12	x	/	/			
Mammals								
*Castor fiber**	120	0.002		x				x
Galemys pyrenaicus	200	13.89						x
*Lutra lutra**	120	0.00017						x
Microtus cabrerae	200	57.5			x			
Microtus oeconomus arenicola	200	65	/		/	/	/	
Microtus oeconomus mehelyi	200	65	/		/	/	/	
Mustela lutreola	200	0.083			/	x	/	
*Myotis capaccinii**	200	0.0042						x
*Myotis dasycneme**	200	0.0042					x	

Appendix B

Table I-A2: Allocation of species to countries
(scenario: coordinated conservation planning within countries)

Each species is allocated to the country in which most occupied planning units of the species are located; the twelve resulting countries encompass 79% of the considered land area.

European country	Species
Bulgaria	*Triturus karelinii, Pelecanus crispus, Pelecanus onocrotalus, Phalacrocorax pygmaeus, Plegadis falcinellus, Tadorna ferruginea*
Finland	*Asio flammeus, Philomachus pugnax*
France	*Bombina variegata, Discoglossus montalentii, Emys orbicularis, Alcedo atthis, Ixobrychus minutus minutus, Milvus migrans, Nycticorax nycticorax, Mustela lutreola*
Germany	*Triturus cristatus, Myotis dasycneme*
Greece	*Mauremys caspica, Hoplopterus spinosus*
Hungary	*Platalea leucorodia, Microtus oeconomus mehelyi*
Italy	*Discoglossus sardus, Pelobates fuscus insubricus, Rana latastei, Salamandrina terdigitata, Triturus carnifex, Elaphe quatuorlineata, Myotis capaccinii*
The Netherlands	*Microtus oeconomus arenicola*
Poland	*Bombina bombina, Triturus montandoni, Acrocephalus paludicola, Aythya nyroca, Botaurus stellaris stellaris, Chlidonias niger, Ciconia ciconia, Ciconia nigra, Crex crex, Haliaeetus albicilla,Porzana parva parva, Porzana porzana, Sterna albifrons, Castor fiber*
Romania	*Triturus dobrogicus, Triturus vulgaris ampelensis, Aquila clanga, Ardeola ralloides, Chlidonias hybridus*
Spain	*Alytes muletensis, Chioglossa lusitanica, Discoglossus galganoi, Mauremys leprosa, Aquila chrysaetos, Ardea purpurea purpurea, Fulica cristata, Gelochelidon nilotica, Glareola pratincola, Marmaronetta angustirostris, Oxyra leucocephala, Porphyrio porphyrio, Porzana pusilla, Galemys pyrenaicus, Lutra lutra, Microtus cabrerae*
Sweden	*Anser erythropus, Gavia arctica, Grus grus, Pandion haliaetus, Tringa glareola*

Appendix C

Table I-A3: Allocation of species to biogeographical regions (scenario: coordinated conservation planning within biogeographical regions)

Each species is allocated to the biogeographical region in which most occupied planning units of the species are located; the seven resulting regions encompass 99% of the considered land area.

Biogeographical region	**Species**
Alpine	*Triturus montandoni, Anser erythropus*
Atlantic	*Chioglossa lusitanica, Microtus oeconomus arenicola, Mustela lutreola*
Black Sea	*Pelecanus onocrotalus*
Boreal	*Asio flammeus, Gavia arctica, Grus grus, Pandion haliaetus, Philomachus pugnax, Tringa glareola, Castor fiber*
Continental	*Bombina bombina, Bombina variegata, Pelobates fuscus insubricus, Rana latastei, Triturus carnifex, Tritutus cristatus, Triturus karelinii, Triturus vulgaris ampelensis, Acrocephalus paludicola, Alcedo atthis, Aquila clanga, Ardeola ralloides, Aythya nyroca, Botaurus stellaris stellaris, Chlidonias niger, Ciconia ciconia, Ciconia nigra, Crex crex, Haliaeetus albicilla, Ixobrychus minutus minutus, Milvus migrans, Pelecanus crispus, Phalacrocorax pygmaeus, Porzana parva parva, Porzana porzana, Sterna albifrons,Lutra lutra, Myotis dasycneme*
Mediterranean	*Alytes muletensis, Discoglossus galganoi, Discoglossus montalentii, Discoglossus sardus, Salamandrina terdigitata, Elaphe quatuorlineata, Emys orbicularis, Mauremys caspica, Mauremys leprosa, Aquila chrysaetos, Ardea purpurea purpurea, Chlidonias hybridus, Fulica cristata, Gelochelidon nilotica, Glareola pratincola, Hoplopterus spinosus, Marmaronetta angustirostris, Nycticorax nycticorax, Oxyra leucocephala,Plegadis falcinellus, Porphyrio porphyrio, Porzana pusilla, Tadorna ferruginea,Galemys pyrenaicus, Microtus cabrerae, Myotis capaccinii*
Pannonian	*Triturus dobrogicus, Platalea leucorodia, Microtus oeconomus mehelyi*

II

Integrating land market feedbacks into conservation planning – a mathematical programming approach

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Abstract. Protected areas have often been designated ad hoc. Despite increasing conservation efforts, loss of biodiversity is still accelerating. Considering land scarcity and demand for alternative uses, efficiency in conservation strongly correlates with efficiency in land allocation. Systematic conservation planning can effectively prioritize conservation activities. Previous studies minimize opportunity costs for given conservation targets. However, these studies assume constant marginal costs of habitat protection. We extend this cost minimization approach by also considering a dynamic representation of marginal costs. The more land is allocated to nature reserves, the higher are opportunity costs, i.e. costs of forgone agricultural production. This increase in costs results from changes in the prices of agricultural commodities. We employ a deterministic, spatially explicit mathematical optimization model to allocate species habitats by minimizing opportunity costs for setting aside land for conservation purposes. The model is designed as a mixed integer programming problem and solved with CPLEX. Our results show the need for integrating land market feedbacks into conservation planning. We find that ignoring land rent adjustments can lead to highly cost-ineffective solutions in reserve selection.

Keywords: marginal costs, mathematical optimization model, mixed integer programming, set-covering problem, systematic conservation planning

1 Introduction

Creation of reserves has often been done ad hoc, leading to inefficient allocation of conservation areas [35, 21, 18]. The selection of protected areas is biased towards economically marginal landscapes, which lead to severe underrepresentation of species, habitats, and ecosystems [36, 4]. Furthermore, existing reserves are often too small to support viable populations of wide-ranging species [37, 11]. Thus, despite increasing conservation efforts, biodiversity loss is still accelerating [27, 6].

Considering land scarcity and demand for alternative uses, efficiency in conservation strongly correlates with efficiency in land allocation. Systematic conservation planning can effectively prioritize conservation activities [21, 34, 22]. The set-covering problem detects how to achieve some minimum representation of biodiversity features while minimizing the resources needed [34, 53].

Addressing high competition for land especially in densely human-populated countries, the set-covering problem identifies the least required area. Previous studies minimize the number of reserve sites or their total area for given representation targets of biodiversity features (e.g., [41, 38, 49]). However, finding the minimum area for reservation does not guarantee minimum costs for achieving the respective conservation target. Costs of conservation, just like the distribution of biodiversity, are not spatially homogeneous [3, 7]. Ando et al. [3] show that the cost per conservation site under cost minimization can be less than one-sixth of that under the site-minimizing solution. As marketable land values differ, regional priorities change under cost minimization [8, 32, 29].

The complex issue of conservation costs has received increasing attention within the last decade. Naidoo and Adamowicz [28] argue that these costs may include acquisition costs, management costs, transaction costs, or opportunity costs. A study by Frazee et al. [16] is the first systematic estimate of the costs of conserving the Cape Floristic Region, a globally recognized biodiversity hotspot. James et al. [20] roughly estimate the costs of a global reserve network. Land acquisition costs account for the largest cost component in their study. Carwardine et al. [12] identify priority areas in Australia for alternative conservation actions including land acquisition and stewardship.

Opportunity cost data can be applied to identify sites that minimize conflicts of alternative uses of land or marine areas while achieving conservation objectives [30]. Stewart et al. [47] employ data from commercial rock lobster fishery to minimize forgone fishing income in a marine reserve system in South Australia. The objective of a study by Faith et al. [14] is to reach conservation goals while reducing forgone opportunities for timber production in Papua New Guinea. Adams et al. [1] argue that an understanding of the spatial distributions of opportunity costs disaggregated to groups of stakeholders is necessary for decision-making on priority conservation areas.

Most studies estimating land acquisition or opportunity costs for conservation assume that land prices do not change regardless of the amount of land allocated to priority areas for conservation. This assumption of constant marginal land costs neglects land market effects and thereby may lead to underestimations of the real costs and thus non-optimal decisions on spatial conservation prioritization. Naidoo et al. [29] argue that setting aside land for conservation itself could change land costs. Armsworth et al. [5] explicitly consider land market feedbacks with respect to conservation planning. Their analysis confirms that land markets may influence conservation efforts even at local scales.

This study demonstrates a method to integrate land market feedbacks directly and consistently into conservation planning. We employ a deterministic, spatially explicit mathematical programming model, which allocates species habitats by minimizing total costs for setting aside land for conservation purposes. We apply mixed integer programming techniques. To illustrate the effect of incorporating the dynamic nature of opportunity costs into conservation planning, we compare different cost representations in a multiple-species conservation planning exercise. We discuss the different degrees of errors that may result from keeping land prices constant. The empirical model application to European wetland biodiversity covers 69 wetland species across 23 European countries.

2 Methods

2.1 Conservation target: integrating representation and persistence

Effective biodiversity conservation requires simultaneous consideration of representation and persistence conditions [21, 42]. In our model, each species is subject to representation targets. These targets can differ across species. We assume the persistence criterion to be fulfilled when two conditions are met. First, each individual species' representation corresponds to one viable population. A population is considered viable when the allocated land area meets the minimum critical area, which is a species-specific measure based on density data and minimum viable population sizes. To account for different habitat quality [15, 39] and potential bias in sampling effort [45], we solve the model for different density data. We do not explicitly portray competition between species and assume that they do not affect each other in terms of density. The second condition for the persistence criterion refers to habitat type requirements. In our model, each species requires specific habitat types which are either necessary for the species' survival or optional habitats. The land area that corresponds to the minimum critical area of a species is allocated to the relevant habitat types.

2.2 Planning units

We use a spatially explicit model based on planning units that differ in shape and size. Planning units are the spatial entities for which species occurrence data exist. We assume constant habitat suitability for a species across all possible planning units. The potential reserve areas are determined for each planning unit. Parts of planning units necessary to fulfill conservation targets are selected as priority area for conservation. If a species' minimum area requirement cannot be fulfilled within a single planning unit, the model selects further habitat area in adjacent planning units. This approach differs from previous conservation planning studies where either total planning units (e.g., [49, 53, 52]) or fractions of them (e.g., [10]) are chosen. Rationale for our method is to overcome the problem of scale difference between grid dimension and land area available for conservation purposes. We have designed our model for relatively large planning units or planning units lying within densely human-populated regions. First, it is unlikely or even impossible to reserve such planning units entirely. Second, species' habitat size requirements will regularly not correspond to the extent of a large planning unit. Marianov et al. [23] present a method to select reserves for species with differential habitat size needs exceeding planning units' areas. Our model additionally acknowledges the fact that area requirements may be smaller than a planning units' area. The total area selected as priority area for conservation in a planning unit includes the minimum critical areas of all species protected in it.

2.3 Land market feedbacks and marginal costs

Most studies estimating opportunity costs for conservation assume that land prices do not change regardless of the amount of land allocated to priority areas for conservation (e.g., [3, 32, 48]). Total land costs are derived by simply multiplying the demanded hectares of land area by current land rents or prices. As indicated by Naidoo et al. [29] and Armsworth et al. [5], this exogenous determination of marginal costs of land as being constant neglects land market feedbacks and hence may underestimate total costs of reservation.

When purchasing or renting large areas for conservation, the equilibrium between supply and demand in regional land markets is distorted and land rental rates will adjust. This feedback from land markets affects the economic feasibility of conservation as well as the costs of future conservation efforts. The more land is allocated to reserves, the higher are its opportunity costs, i.e. costs of forgone agricultural production. This increase in opportunity costs results from price adjustments in agricultural commodity markets. To consider land market feedbacks in conservation planning, the land rent dynamics need to be represented endogenously in reserve selection models.

The rent for an additional hectare of land represents the marginal cost of land. Mathematically, the marginal cost function is expressed as the derivative of the total cost function with respect to quantity. According to economic theory, a competitive land supply curve is equal to the marginal cost function of land. In this study, we employ a linear land supply function. The price elasticity of supply measures the responsiveness of land supply to a change in land rent. Mathematically, the price elasticity of supply is given by (1). Our study shows and compares both exogenous and endogenous representations of land costs (Figure II-1).

$$\varepsilon_{\text{land supply, land rent}} = \left| \frac{\text{\% change in land supply}}{\text{\% change in land rent}} \right| \tag{1}$$

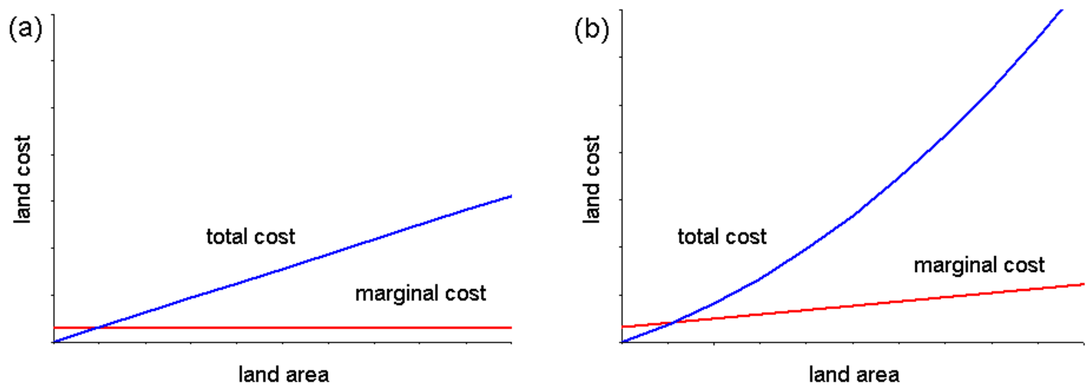

Figure II-1: Exogenous (a) and endogenous (b) representation of land costs

2.4 *Mathematical model structure*

The formal framework utilized here expands the set-covering problem. We use the following notation: $c = \{1,\ldots,C\}$ is the set of countries; $p = \{1,\ldots,P\}$ is the set of planning units; $t = \{1,\ldots,T\}$ is the set of habitat types; $q = \{1,\ldots,Q\}$ is the set of habitat qualities; and $s = \{1,\ldots,S\}$ is the set of species. We employ several set mappings, which contain possible combinations between two or more individual indexes. In particular, $u(s,t)$ identifies the mapping between species and required or optional habitat types and $k(s,p,t)$ possible existence of species and habitats in each planning unit. The objective variable O represents total opportunity costs. The variable Z_c represents opportunity cost per country c. The variable $Y_{p,t,q}$ determines the habitat area per planning unit p, habitat type t, and habitat quality q in hectares. $X_{s,p}$ is a binary variable with $X_{s,p} = 1$ indicating species s is represented in planning unit p, and

$X_{s,p} = 0$ otherwise. r_c denotes the annual land rent per hectare and country c. $a_{p,t,q}$ contains the maximum available area per planning unit p, habitat type t and habitat quality q. $d_{s,q}$ represents species- and habitat quality-specific density data. m_s is a species-specific proxy for minimum viable population size. $h_{t,s}$ determines which habitat types t are required by species s. t_s is the representation target per species s. v_s specifies deviations from the representation target based on exogenous maximum occurrence calculations.

I Conservation planning with constant (exogenous) land rents

Minimize $O = \sum_c Z_c$ (2)

subject to:

$$Z_c = r_c \cdot \sum_{p \in c,t,q} Y_{p,t,q} \qquad \text{for all c} \qquad (3)$$

$$Y_{p,t,q} \le a_{p,t,q} \qquad \text{for all p,t,q} \qquad (4)$$

$$\sum_p X_{s,p} \ge t_s - v_s \qquad \text{for all s} \qquad (5)$$

$$\sum_q Y_{p,t,q} \ge h_{t,s} \cdot X_{s,p} \qquad \text{for all p,t,s} \qquad (6)$$

$$\sum_{t,q} d_{s,q} \cdot Y_{p,t,q} \Big|_{k(s,p,t) \wedge u(s,t)} \ge m_s \cdot X_{s,p} \qquad \text{for all p,s} \qquad (7)$$

$$\sum_{p,t,q} d_{s,q} \cdot Y_{p,t,q} \Big|_{k(s,p,t)} \ge t_s \cdot m_s \qquad \text{for all s.} \qquad (8)$$

The objective function (2) minimizes total costs across all planning units. Equation (3) calculates the total costs per planning unit as product of habitat area and land rent. This formulation displays an exogenous representation of land costs. Constraint (4) limits habitat areas in each planning unit to given endowments. Constraint (5) implements representation targets for all species but allows deviations if the number of planning units with occurrence data is below the representation target. Constraint (6) forces the existence of required habitat types for all species chosen in a particular planning unit. Constraint (7) ensures that the habitat area for the conservation of a particular species is large enough to support viable populations of that species. The summation over habitat types depicts the choice between possible habitat alternatives. Constraint (8) ensures that the total population size equals at least the representation target times the minimum viable population size. This constraint is especially relevant for cases where the representation target is higher than the number of available planning units for conservation. For example, a representation target of ten viable populations with possible species occurrences in only nine planning units would under (8) require one or more planning units to establish enough habitat for more than one viable population.

II Conservation planning with dynamic (endogenous) land rents

To represent land rents endogenously, we alter equation (3) of the model formulation. r^0_c represents the initial land rent per hectare of land and differs by country. $a^0_{p,t,q}$ is the initially available area per planning unit. Land rents r_c rise according to function $f(Y_{p,t,q})$ (9).

$$r_c = f(Y_{p,t,q}) = r_c^0 + b_c \cdot \sum_{p \in c,t,q} Y_{p,t,q} \tag{9}$$

We assume a linear marginal cost function with slope *b*. To determine *b* we introduce different price-elasticities of supply ε at a land supply level equal to the maximum conservation area. The elasticity ε measures the responsiveness of land supply to a change in land rent (10).

$$\varepsilon_c = \left| \frac{\partial \sum_{p \in c,t,q} Y_{p,t,q}}{\partial r_c} \cdot \frac{r_c}{\sum_{p \in c,t,q} a^0_{p,t,q}} \right| = \left| \frac{1}{b_c} \cdot \frac{r_c}{\sum_{p \in c,t,q} a^0_{p,t,q}} \right| \tag{10}$$

The linear marginal cost function $f(Y_{p,t,q})$ is given by (11):

$$f(Y_{p,t,q}) = r_c^0 + \frac{1}{\varepsilon_c} \cdot \frac{r_c}{\sum_{p \in c,t,q} a^0_{p,t,q}} \cdot \sum_{p \in c,t,q} Y_{p,t,q} \tag{11}$$

The corresponding total cost function $F(Y_{p,t,q})$ is (12):

$$F(Y_{p,t,q}) = r_c^0 \cdot \sum_{p \in c,t,q} Y_{p,t,q} + \frac{1}{2\varepsilon_c} \cdot \frac{r_c^0}{\sum_{p \in c,t,q} a^0_{p,t,q}} \cdot \left(\sum_{p \in c,t,q} Y_{p,t,q} \right)^2 \tag{12}$$

In the model formulation, we replace equation (3) with (3a):

$$Z_c = r_c^0 \cdot \sum_{p \in c,t,q} Y_{p,t,q} + \frac{1}{2\varepsilon_c} \cdot \frac{r_c^0}{\sum_{p \in c,t,q} a^0_{p,t,q}} \cdot \left(\sum_{p \in c,t,q} Y_{p,t,q} \right)^2 \quad \text{for all c.} \tag{3a}$$

The model is programmed in General Algebraic Modeling System (GAMS) software version 22.9. It is solved with a mixed integer programming algorithm from CPLEX version 12.1.

3 Application to European wetland biodiversity

3.1 *Ecological and spatial data*

Due to their relevance for conservation and related environmental objectives, we apply our model to freshwater wetlands. Species dependent on freshwater wetlands serve as surrogates for biodiversity. We include 69 tetrapod species listed in the appendices of the Birds and the Habitats directive (79/409/EEC; 92/43/EEC) which encompass 15 amphibian, 4 reptile, 41 breeding bird, and 9 mammal species. Recorded occurrences from species atlases [19, 17, 26] identify their potential distribution in Europe. Species' density data were compiled through literature review; we use the maximum observed density. Proxies for minimum viable population sizes are based on Verboom et al. [50]. We adapt their proposed standards for minimum population sizes depending on species' body sizes and life expectancy. Specifically, a viable population in our model requires 120 reproductive units (pairs/territories/families; depending on species group) of long-lived or large vertebrates and 200 reproductive units of other vertebrates. Data on habitat type requirements are also taken from the literature. We include five broad wetland habitat types in our dataset, namely mires, wet forests, wet grassland, water courses, and water bodies. A further type "open water" is applied to species that require either water courses or water bodies. See the Online Resource for the ecological data included for the 69 species.

Geographically estimated data from Schleupner [43] provide information on existent habitat areas in Europe. To enable the most area-demanding species to fulfill their area requirements, they are allowed to inhabit a certain share of non-wetland habitat. The dataset comprises the European Union with 23 out of 27 member states (see Figure II-2). We excluded Cyprus, Malta, the new member states Romania and Bulgaria, and the Portuguese and Spanish islands in the Atlantic Ocean due to data deficiencies. The planning units coincide with the resolution of the species occurrence data. The atlases use the Universal Transverse Mercator (UTM) projection with grid squares of about 50 km edge length. We considered the terrestrial parts of all 1996 grid cells belonging to the selected European countries as planning units.

Figure II-2: Spatial scope of the empirical model application

3.2 Economic data

We use country-specific data on current agricultural land rents from European land statistics as the opportunity costs for conservation (see Table II-1). Employing only one cost statement per country is undoubtedly a simplification of the distribution of real costs. Bladt et al. [10] present, for example, a method to derive fine scale costs for European conservation planning. However, in order to demonstrate the effect of dynamic cost representations on conservation plans, a simple cost stratification seems appropriate.

To address the uncertainty of the price elasticity of additional land supply, we use elasticities of 0.1, 0.3, and 0.5. These elasticity coefficients portray a price inelastic supply of land. In future studies, we plan to take land prices directly from the European Forest and Agricultural Sector Optimization Model [44].

Table II-1: Agricultural land area and rents for European countries

	Rent for agricultural land [€/ha*a][a]	Agricultural land area [Mha][b]	Total land area [Mha][b]
Austria	244.53	3,240	8,245
Belgium	151.76	1,370	3,028
Czech Republic	23.17	4,249	7,725
Denmark	315.00	2,663	4,243
Estonia	15.76	823	4,239
Finland	152.08	2,295	30,409
France	109.35	29,418	54,766
Germany	156.32	16,950	34,877
Greece	402.98	8,280	12,890
Hungary	54.56	5,807	8,961
Ireland	212.76	4,276	6,889
Italy	248.42	13,888	29,414
Latvia	8.34	1,839	6,225
Lithuania	17.14	2,695	6,268
Luxembourg	150.38	131	259
The Netherlands	396.01	1,914	3,376
Poland	68.08	16,177	30,425
Portugal	158.51	3,496	9,150
Slovakia	13.33	1,930	4,810
Slovenia	86.21	500	2,014
Spain	145.40	28,660	49,898
Sweden	98.12	3,136	41,033
United Kingdom	190.34	17,647	24,193
		171,384	**383,337**

[a] data derived from Eurostat (averaged data from 1985 to 2006 for Austria, Belgium, Denmark, Finland, France, Germany, Greece, Hungary, Ireland, Lithuania, Luxembourg, The Netherlands, Poland, Slovakia, Spain, Sweden, United Kingdom) and Farm Accountancy Data Network (FADN) (data from 2004 for Czech Republic, Estonia, Italy, Latvia, Portugal, Slovenia)

[b] data taken from FAOSTAT (data from 2007)

3.3 *Empirical Results*

Figure II-3 shows annual land opportunity costs for conservation targets ranging from 1 to 25 for 69 wetland species. The model simulations with constant exogenous land rents result in substantially lower total costs when compared to simulations that include land market feedbacks. For the medium price-elasticity of land supply (ε=0.3), the endogenously determined costs are on average about 19 percent (range: 2.0 to 27.9 percent) higher than the exogenously calculated costs. Note, however, that the cost differences in Figure II-3 only represent a fraction of the total cost error resulting from incorrect assumptions about land markets. Specifically, the distance between the individual lines identifies the minimum cost error for incorrectly specified land rents.

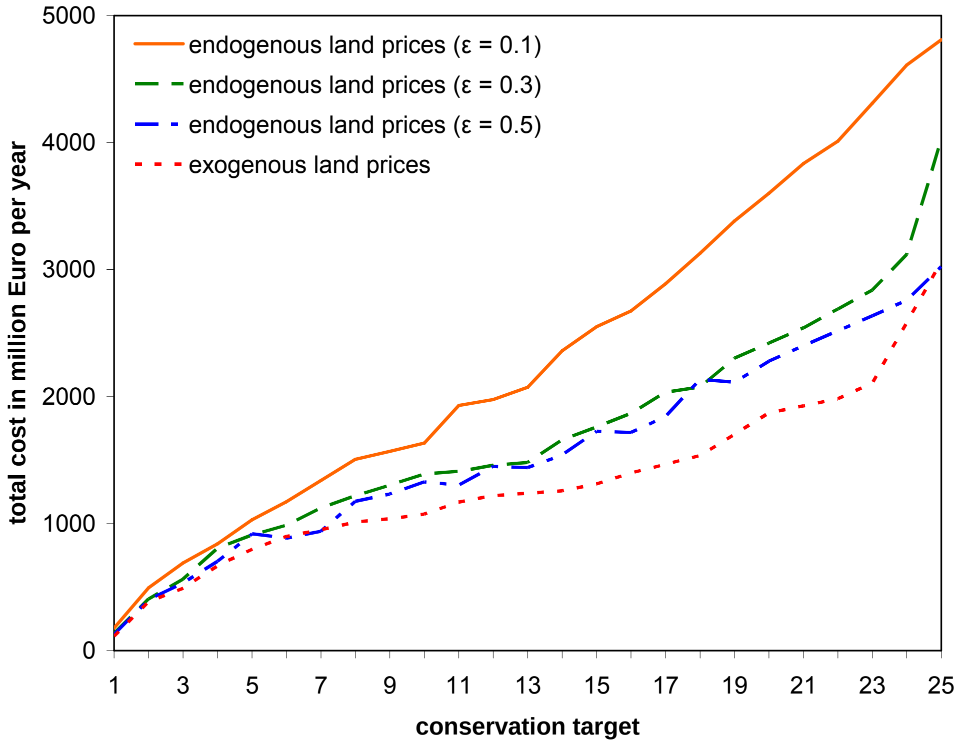

Figure II-3: Total costs resulting from exogenous and endogenous cost representations

The true cost error is likely to be higher because misspecified land rents are likely to result in inefficient land allocations. This is illustrated in Figure II-4, where we correct the land opportunity costs estimated under constant land rents (lower line) to account for land market feedbacks (upper line). We re-calculate national land rents but keep the size and locations of the conservation areas as determined under the setup with constant land rents. Note that the resulting cost function (upper line) is about three to five times higher in magnitude than the endogenous land rent based cost function for the same elasticity (middle line). This indicates that the cost error due to inefficient land allocation may be substantial, especially if there is a large heterogeneity in land prices (see Table II-1).

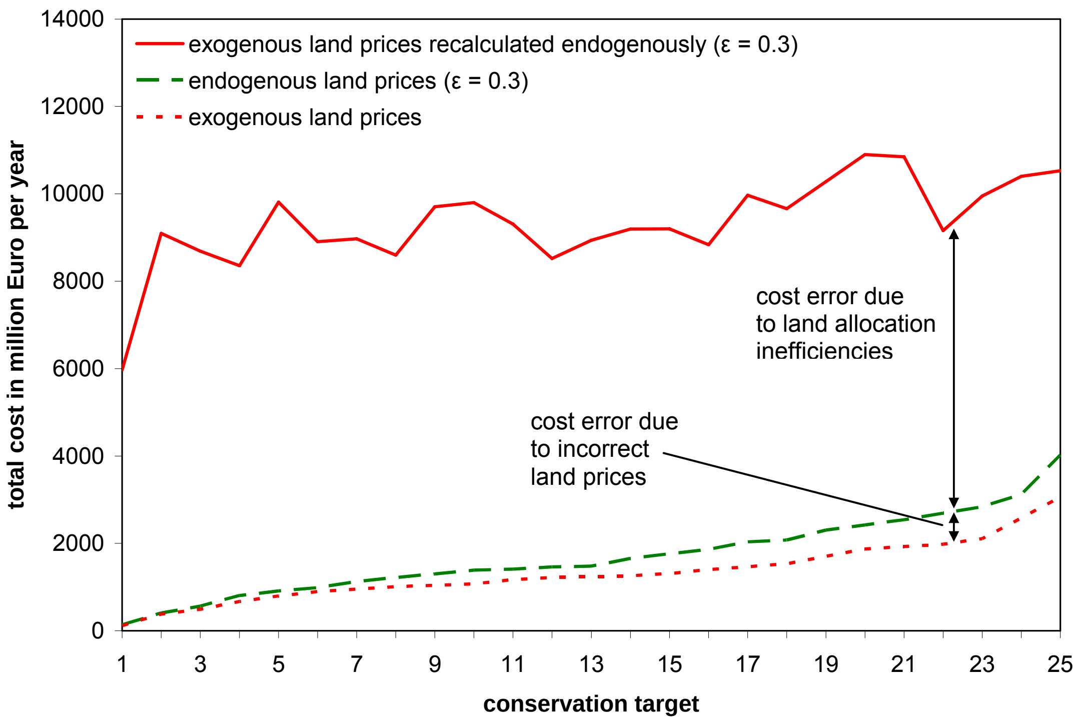

Figure II-4: Cost errors related to exogenous land prices

Figure II-5 shows the total area requirements for achieving a given conservation target and the corresponding habitat shares. For the highest simulated conservation target of 25 viable populations, the total habitat requirements equal 35 to 40 million hectares. This value is about 10 percent of the terrestrial land area and about 20 percent of the current agricultural area of the considered countries (see Table II-1). Furthermore, the comparison of Figure II-5a and b reveals that different assumptions about land rents have little impact on the total conservation area requirements. However, the regional reserve allocation between European countries differs across alternative land market representations (Figure II-6).

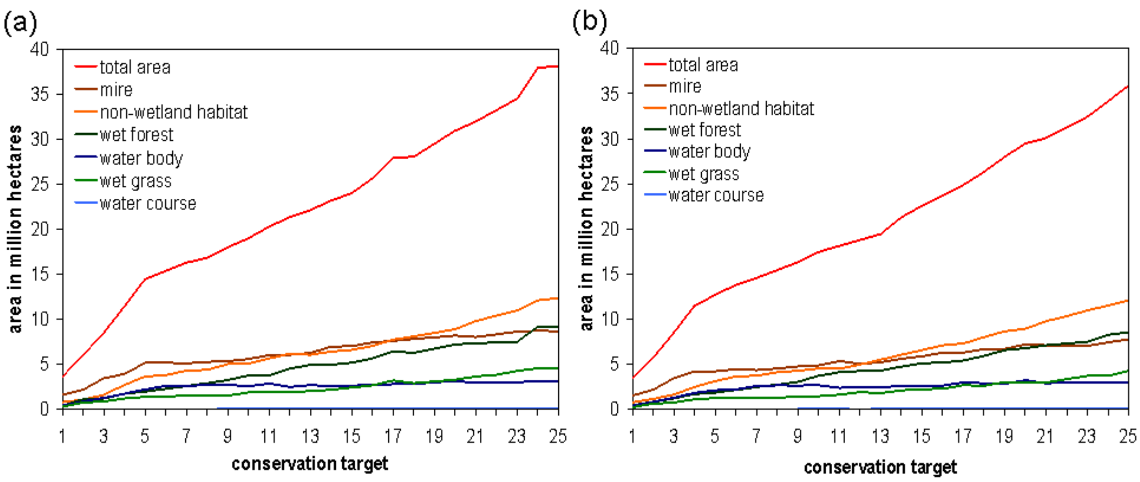

Figure II-5: Allocation to wetland habitat types and total area requirement: exogenous (a) and endogenous (b) (ε=0.3) land prices

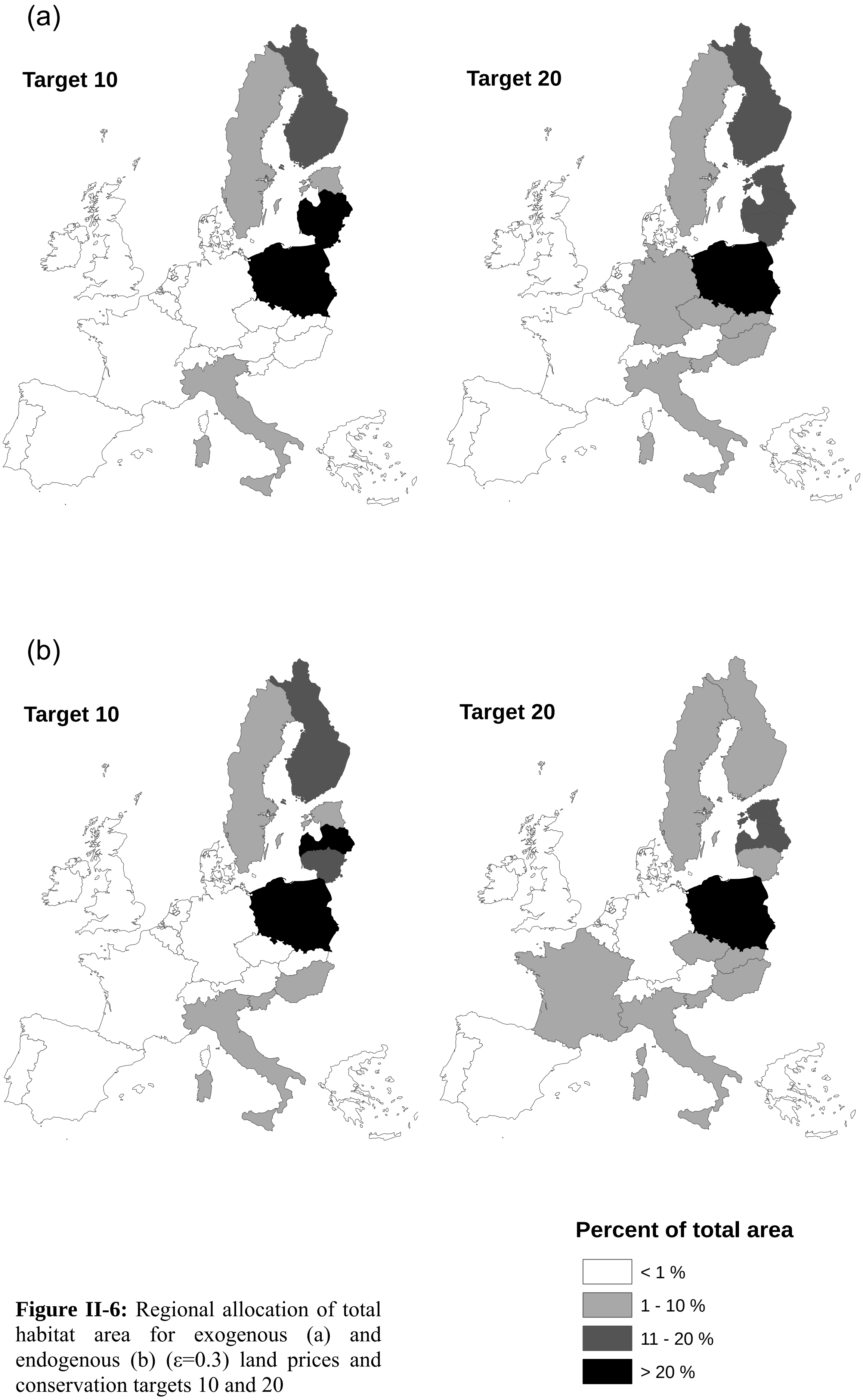

Figure II-6: Regional allocation of total habitat area for exogenous (a) and endogenous (b) (ε=0.3) land prices and conservation targets 10 and 20

4 Discussion

Conservation is costly and available land resources are scarce. Applying economic concepts and tools becomes increasingly important for decision-making in conservation [46, 29, 51]. However, when appropriate data on land values are not available, conservation planning studies often use area as a proxy for costs [25]. As Cullen et al. [13] point out, failure to apply economic tools to decision-making in conservation problems may lead to errors in project selection, wasteful use of scarce resources, and lower levels of conservation than could potentially be achieved from the given resources. The review of reserve designs by Newburn et al. [31] finds that land costs are often inadequately considered.

In order to effectively allocate scarce conservation funds, a full integration of economic costs into spatial conservation prioritization is inevitable. This necessarily includes the consideration of land market feedbacks and marginal costs. Our study shows that negligence of land market adjustments may lead to highly cost-ineffective reserve selection. Furthermore, the reported total costs would be misleading and substantially underestimate the true total costs.

Polasky et al. [33] argue that assuming constant land prices is reasonable when the areas of conservation interest do not significantly impact agricultural and forestry commodity markets. However, the need for nature reserves does not just regard a few local sites for a few species. Most of the demand can only be met by reverting a considerable portion of agricultural and managed forest sites back to nature areas. Since virtually all agricultural and forest production is directly or indirectly linked to regional and international commodity markets, there will always be a market feedback. Armsworth et al. [5] confirm that land market feedbacks can influence conservation efforts even at local scales.

A meaningful integration of opportunity costs into reserve selection models requires reliable data on land rents and price-elasticity of land. These economic parameters not only enable the implementation of land market feedbacks in conservation planning but also facilitate possible linkages between reserve selection models and integrated land use models for the simultaneous economic and environmental assessment of land use options. Those models (e.g., FASOM [2], ASMGHG [24]) analyze environmental impacts such as greenhouse gas emissions, water quality, or soil erosion from the adoption of land use strategies. They so far neglect biodiversity conservation as an explicit land use option.

Several important simplifications on economic and ecological issues in our analysis need to be noted. First, we consider only the land opportunity costs from acquiring additional land and keeping existing land under conservation. Reality in conservation planning is more complex and there are important additional costs, i.e. costs related to reserve establishment and maintenance [29]. Note that opportunity costs are also relevant in other conservation issues not included in our study. In some cases, management practices of landowners change and are compensated for.

For example, Barlow et al. [9] estimate the foregone forestry potentials when managing forest for maintaining the habitat of endangered species. Rondinini and Boitani [40] analyze the costs of antipredator measures associated with the conservation of large carnivores. Second, we do not account for spatial reserve design criterions such as connectivity or compactness in our model and we do not consider spatio-temporal aspects of persistence. We apply only five coarse habitat classes with no quality differences. Note that the employed absolute values of species' pairs, territories or families serve as proxies for viable populations. They are not assumed to represent real minimum viable populations, but serve as working targets due to the lack of better data.

5 Implications for conservation planning

Biodiversity conservation is a declared objective of national governments but also of the United Nations. Its realization may interfere with other objectives because of land competition between conservation areas, agricultural fields, bioenergy plantations, and intensively managed forests. Our study quantifies the cost implications of different conservation planning approaches for 69 species of European wetlands. We find that misspecified land markets may lower the reported cost estimates but increase the true costs of conservation by several orders of magnitude. Depending on how an incorrectly specified conservation planning study is used, there are different degrees of errors. Moderate errors occur if a conservation assessment with misspecified land rents is used only to predict the costs of conservation efforts. In this case, the realization of the conservation plan will result in higher costs or reduced areas. However, if an incorrectly specified assessment also serves to determine the optimal locations for conservation areas, additional costs arise from inefficient reserve allocations. Considering land market feedbacks seems particular important in cases where (i) land rents or prices are comparably high, (ii) high competition for land occurs, or (iii) a great fraction of land is to be reserved within a region.

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Supplementary Material

Table II-A1: Wetland species of European conservation concern: Minimum viable population sizes, densities, and habitat types

Scientific name	Minimum viable population size[3] [reproductive units]	Maximum density[4,5] [reproductive units/hectare]	Required (x) and optional (/) habitat types					
			Mire	Wet forest	Wet grassland	Water course	Water body	Open water[6]
Amphibians								
Alytes muletensis	200	20				x		
Bombina bombina	200	20			x		x	
Bombina variegata	200	20		/	/		x	
Chioglossa lusitanica	200	10				x		
Discoglossus galganoi[1]	200	10					x	
Discoglossus montalentii	200	10				x		
Discoglossus sardus	200	10					x	
Pelobates fuscus insubricus	200	10					x	
Rana latastei	200	20		x			x	
Salamandrina terdigitata	200	10				x		
Triturus carnifex	200	10		/	/		x	
Triturus cristatus	200	10		/	/		x	
Triturus dobrogicus	200	10			/		x	
Triturus karelini	200	10					x	
Triturus montandoni	200	10		x	/		x	
Triturus vulgaris ampelensis	200	20		/	/		x	
Reptiles								
Elaphe quatuorlineata	120	2			/			
Emys orbicularis	120	15					x	
Mauremys caspica	120	9						x
Mauremys leprosa	120	9						x
Birds								
Acrocephalus paludicola	200	1.09			x			
Alcedo atthis	200	0.15						x
Anser erythropus	200	0.127		x				x
*Aquila chrysaetos**	120	0.0002	/		/			
*Aquila clanga**	120	0.000055	/	x	/	/	/	
Ardea purpurea purpurea	120	0.19			x			x
Ardeola ralloides	200	0.19			x		x	
Asio flammeus	200	0.1	/		/			
Aythya nyroca	200	1			x		x	
Botaurus stellaris stellaris	200	0.5			x			
Chlidonias hybridus	200	0.19			/		x	
Chlidonias niger	200	0.19			x		x	
*Ciconia ciconia**	120	0.001415			x			*x*
*Ciconia nigra**	120	0.00018		x				x
Crex crex	200	0.19	/		x	/		
Fulica cristata	200	10			x		x	

Scientific name	Minimum viable population size[3] [reproductive units]	Maximum density[4,5] [reproductive units/hectare]	Required (x) and optional (/) habitat types					
			Mire	**Wet forest**	**Wet grassland**	**Water course**	**Water body**	**Open water[6]**
Gavia arctica	120	0.006					x	
Gelochelidon nilotica	200	0.19			x	x		
Glareola pratincola	200	8			x		x	
*Grus grus**	120	0.00043	/	/	/		/	
Haliaeetus albicilla	120	0.01273		x				x
Hoplopterus spinosus	200	0.3846			x			x
Ixobrychus minutus minutus	200	1.97			x			x
Marmaronetta angustirostris	200	0.19			x		x	
Milvus migrans	120	1.2733						x
Nycticorax nycticorax	200	0.19			x			x
Oxyura leucocephala	200	1.5					x	
*Pandion haliaetus**	120	0.0004		/			x	
Pelecanus crispus	120	0.19			/		x	
Pelecanus onocrotalus	120	0.19			/		x	
Phalacrocorax pygmaeus	200	0.19		/	/		x	
Philomachus pugnax	200	1	/		/			
Platalea leucorodia	120	0.19		/	x		x	
Plegadis falcinellus	200	0.19		/	x		x	
Porphyrio porphyrio	200	3.3			x		x	
Porzana parva parva	200	5			x		/	
Porzana porzana	200	0.333	/		/			
Porzana pusilla	200	3.5368			x			
Sterna albifrons	200	0.19				x	/	
Tadorna ferruginea	120	10					x	
Tringa glareola	200	0.12	x	/	/			
Mammals								
Castor fiber[2,]*	120	0.002		x				x
Galemys pyrenaicus	200	13.89						x
*Lutra lutra**	120	0.00017						x
Microtus cabrerae	200	57.5			x			
Microtus oeconomus arenicola	200	65	/		/	/	/	
Microtus oeconomus mehelyi	200	65	/		/	/	/	
Mustela lutreola	200	0.083			/	x	/	
*Myotis capaccinii**	200	0.0042						x
*Myotis dasycneme**	200	0.0042					x	

[1] including *Discoglossus jeanneae*

[2] except the Estonian, Latvian, Lithuanian, Finnish and Swedish populations (according to 92/43/EEC)

[3] adapted from Verboom et al. (2001)

[4] densities colonial birds: distinction in nesting and foraging area; foraging area is set to 5 ha per reproductive unit

[5] densities amphibians: 10 reproductive units per hectare for solitary species; 20 reproductive units per hectare for gregarious species

[6] open water: water course or water body

* area-demanding species that are allowed to inhabit up to 50% non-wetland habitat

III

Benefits of global earth observation for conservation planning in the case of European wetland biodiversity

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Abstract. This study investigates benefits of improved land cover and land value information for biodiversity protection. We apply a habitat allocation model that is based on principles from systematic conservation planning and economic theory. It estimates area requirements and opportunity costs of habitat protection for the European continent simultaneously covering endangered wetland species and corresponding habitat types. The model is solved for a range of biodiversity targets and conservation scenarios. We compare the impacts of employing non-GEOSS vs. GEOSS data based simulations where two alternative resolutions are used for two input datasets. First, habitat locations are either restricted only by historical species occurrence data at a UTM 50 resolution or by explicit wetland data at a 1 km² resolution. Second, coarse country-average land rents are contrasted with spatially detailed land rent estimates at a 5' resolution. Results show that the accuracy of conservation plans improves considerably with higher resolution habitat data and spatially explicit land rent data. Particular benefits include improved estimations on area requirements and opportunity costs of habitat protection for given conservation targets and improved regional allocation plans for conservation areas to ensure adequate site quality and species coverage. In our application, misspecifications of conservation costs due to coarse scale land rent data range from -14.7 to +3.9 percent. Inadequate habitat data result in notable reductions in conservation target achievement.

Keywords: systematic conservation planning, biodiversity policy, land use optimization model, mixed integer mathematical programming, spatial wetland distribution, global earth observation data resolution, homogenous response units

1 Introduction

Global earth observation (GEO) is fundamental to achieve sustainable development (Group on Earth Observations 2005). Recent studies have started to look at the benefits of Earth Observation (see for example Pricewaterhouse Coopers, 2006). However, there have been no comprehensive assessments of their economic, social and environmental benefits to date. A vision is to develop a high quality, timely, and comprehensive Global Earth Observation System of Systems (GEOSS). This includes a global biodiversity observation system that accommodates the data needs of national governments, monitoring bodies for international environmental agreements, natural resource planners, scientific researchers and civil society. A GEOSS biodiversity observation system would create a mechanism to integrate biodiversity data with other observations more effectively, leverage investments in local and national research and observation projects and networks for global analysis and modelling. It could build on existing efforts and collectively provide essential data and models for monitoring and reporting in the framework of the biodiversity-related conventions, and provide new information and tools for biodiversity research (Group on Earth Observations 2005). To evaluate the status of biodiversity and to determine how current conservation efforts can be improved, biodiversity monitoring is crucial (Balmford *et al.* 2005). For example, there are proposals to establish global biodiversity monitoring systems (Pereira and Cooper 2006, Scholes *et al.* 2008) which include, harmonize, and expand ongoing monitoring activities (Henry *et al.* 2008). To efficiently regulate such efforts, methodologies and analytical tools need to be developed to assess societal benefits of global earth observation (see e.g. Fritz *et al.* 2008).

This study contributes to the benefit assessment of GEO in the realm of biodiversity and ecosystems. In particular, we investigate conservation plans for European freshwater wetlands, recognizing both their vulnerability to human disturbances (Bobbink *et al.*, 2006; Bronmark and Hansson, 2002) and their potential contribution to environmental objectives including among others biodiversity conservation (Mitsch and Gosselink, 1993; Schweiger *et al.*, 2002), carbon storage (Belyea and Malmer, 2004; Zhou *et al.*, 2007), and provision of water-related ecosystem services (Brauman *et al.*, 2007). On the densely populated European continent, competition for land is high. Agricultural expansion and intensification lead to habitat loss, degradation, and fragmentation. These factors are the most important threats to biodiversity (Mace *et al.* 2005). The European Union has made a political commitment to halt the decline of biodiversity by 2010 (Göteborg European Council, 2001). Despite efforts to protect habitats, species, and ecosystems, Europe will not achieve this target (EEA, 2009). Due to land scarcity and demand for alternative uses, efficiency in biodiversity conservation strongly depends on the efficiency in land allocation. Systematic conservation planning provides tools to identify optimally located priority areas for conservation (Margules and Pressey 2000, Possingham *et al.* 2000). However, efficient land allocation is only possible when these tools are used with adequate and reliable data.

An important element of data quality relates to spatial resolution. In this study, we focus on two data categories that are important for wetland biodiversity conservation planning. These include data on the distribution of existing and potential wetland habitat areas and data on land rents. Consistent, adequately resolved data for the geographically distribution of wetland areas in Europe do not exist. The spatial distribution of wetlands in Europe is not well known except for selected large wetland areas or for wetlands of special ecological interest (Merot *et al.* 2003). Furthermore, country statistics differ in spatial accuracy, reliability, acquisition method, and class definition. Aggregating statistical and spatial data from many sources into one database may lead to low spatial accuracy and reduce comparability, i.e. between eastern and western European countries. At present, CORINE (EEA 2000) is the most detailed land cover database for the European Union as a whole. Nevertheless, it only distinguishes functional land use which makes its classes heterogeneous. For ecosystem studies, land cover maps would be more useful. In contrast, the digital map of the potential natural vegetation of Europe (Bohn and Neuhäusel 2003) shows a detailed classification and potential distribution of wetland vegetation types across Europe. However, this distribution does not account for human influences such as river regulation, peat extraction or urbanization, which may substantially impair wetland restoration. To estimate the cost of habitat protection, accurate data on land rents are necessary. European statistics (e.g. Eurostat) and models such as the Global Trade Analysis Project (GTAP) model (Lee *et al.* 2009) provide comprehensive data on land rents. However, these data are not spatially explicit. In this study, we use productivity differences at Homogenous Response Units (HRU, Skalsky *et al.*, 2008) to establish geographically more accurate land rent data.

Obtaining finer scaled GEO data is costly and questions arise if and how much conservation planning will benefit from the availability of better data. In this case study of European wetlands, we consider the impact of methodology and data on land allocation efficiency for biodiversity conservation. We i) employ an adequate conservation planning tool, the HABITAT model. We ii) develop specific high-resolution data on wetland habitats and land rents for Europe to replace frequently used coarse spatial datasets in conservation planning processes. We discuss the different degrees of errors that may result from employing coarse scale data and thereby assess the benefits of GEO data. The empirical model application to European wetland biodiversity covers 72 wetland species across the entire European continent.

2 Methods

We employ HABITAT; a deterministic, spatially explicit, mathematical optimization model that allocates land in order to meet conservation objectives. To establish fine scale input data, we integrate spatially explicit wetland habitat areas and high resolution land rent data for the European continent. Hereafter, we refer to these datasets as “GEOSS data”. The GEOSS data serve to replace frequently used “non-GEOSS data” with a coarser spatial resolution. Subsequently, we use the HABITAT model to simulate conservation policy scenarios to examine the impact of the alternative data on the optimal conservation plan. Figure III-1 illustrates the overall structure of the study.

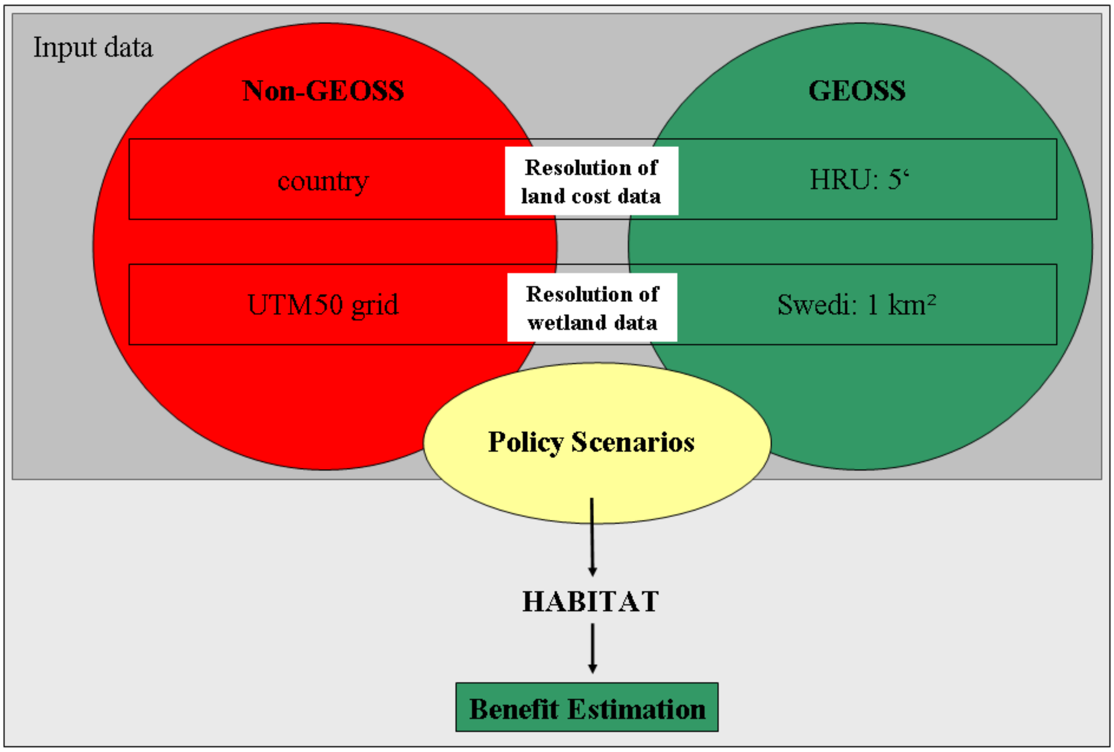

Figure III-1: Overview of the study

2.1 *The habitat allocation model HABITAT*

2.1.1 *Model characteristics and input data*

HABITAT is a deterministic, spatially explicit mathematical optimization model which is programmed in General Algebraic Modeling System (GAMS). The model is solved with a mixed integer programming algorithm from CPLEX version 12.1.

Conceptually, HABITAT depicts the set-covering problem from systematic conservation planning. Its objective is to minimize total resource expenditure, subject to the constraint that all biodiversity features meet exogenously given conservation objectives (Possingham *et al.* 2000, McDonnell *et al.* 2002). Conservation objectives account for the two principal conditions of systematic conservation planning: representation and persistence of the biodiversity features (Margules & Pressey 2000, Sarkar *et al.* 2006). Species are subject to exogenously assigned representation targets. Each representation corresponds to one minimum viable population (MVP) of that species. The land area necessary to sustain a MVP is allocated to habitat types required by that species.

72 wetland vertebrate species of European conservation concern listed in the Birds and the Habitats Directive (79/409/EEC, 92/43/EEC) serve as surrogates for biodiversity in our model. Vertebrate species are common surrogates for biodiversity (Araujo *et al.* 2007, Rodrigues and Brooks 2007, Tognelli *et al.* 2008) as there are relatively good occurrence data available and they usually have greater area demands than invertebrates, plant species, and even most ecosystems. The species assemblage includes 16 amphibians, 4 reptiles, 43 breeding birds, and 9 mammals. Recorded occurrences from Gasc *et al.* (1997), Hagemeijer and Blair (1997), and Mitchell-Jones *et al.* (1999) identify their European distribution.

Population density data for all 72 species are equal to the maximum observed densities from a comprehensive literature review. In addition, we use the proposed standards for minimum population sizes from Verboom *et al.* (2001) as proxies for MVP size. We distinguish five broad wetland habitat types including peatlands, wet forests, wet grasslands, water courses, and water bodies. Information on species' habitat type requirements also result from literature review. See supplementary material A for the ecological data of the 72 wetland species.

HABITAT is a spatially explicit model with many planning units of varying shape and size. The potential habitat area to be selected is specified for each planning unit. There are two possible conservation states indicating whether a planning unit is used as a species' reserve (1) or not (0). Assigning a planning unit as a species reserve is only possible if this species was historically observed in a planning unit or in its close proximity. Parts of planning units necessary to fulfill conservation targets are selected as reserves. If species' area requirements cannot be fulfilled within a single planning unit, further habitat is selected in adjacent planning units. This procedure allows easy implementation of planning units with varying sizes.

The resolution of the planning units is consistent with that of the species occurrence data. The Universal Transverse Mercator (UTM) projection results in grid squares of about 50 km edge length. The terrestrial parts of all 2725 grid cells encompassing the whole European continent serve as planning units. Cyprus, Malta, and Macaronesia are excluded from the analysis due to species data deficiencies.

2.1.2 Mathematical model structure

We use the following notation: $c = \{1,\ldots,C\}$ is the set of countries; $p = \{1,\ldots,P\}$ is the set of planning units; $t = \{1,\ldots,T\}$ is the set of habitat types; $q = \{1,\ldots,Q\}$ is the set of habitat qualities; $s = \{1,\ldots,S\}$ is the set of species. We employ several set mappings, which contain possible combinations between two or more individual sets. In particular, $u(s,t)$ identifies the mapping between species and habitat types and $k(s,p,t)$ the possible existence of species and habitats in each planning unit. The objective variable O represents total opportunity costs. The non-negative variable array Z_c represents opportunity cost in country c. Another non-negative variable array $Y_{p,t,q}$ depicts the habitat area for planning unit p, habitat type t, and habitat quality q in hectares. $X_{s,p}$ is a binary variable array with $X_{s,p} = 1$ indicating species s is represented in planning unit p, and $X_{s,p} = 0$ otherwise. The model's exogenous data are given in small italic letters. $r_{c,p}$ denotes the annual land rent per hectare in country c and planning unit p. $a_{p,t,q}$ contains the maximum available area for planning unit p, habitat type t and habitat quality q. $d_{s,q}$ represents species- and habitat quality-specific population density data. m_s is a species-specific proxy for the minimum viable population size. $h_{t,s}$ determines non-substitutable habitat requirements for habitat type t and species s. t_s is the desired representation target for species s. v_s specifies possible deviations from the representation target based on exogenously calculated occurrence maxima.

Minimize $$O = \sum_c Z_c \qquad [1]$$

subject to:

$$Z_c = \sum_{p,t,q} Y_{p,t,q} \cdot r_{c,p} \qquad \text{for all c} \qquad [2]$$

$$Y_{p,t,q} \le a_{p,t,q} \qquad \text{for all p,t,q} \qquad [3]$$

$$\sum_p X_{s,p} \ge t_s - v_s \qquad \text{for all s} \qquad [4]$$

$$\sum_q Y_{p,t,q} \ge h_{t,s} \cdot X_{s,p} \qquad \text{for all p,t,s} \qquad [5]$$

$$\sum_{t,q} d_{s,q} \cdot Y_{p,t,q}\Big|_{k(s,p,t)\wedge u(s,t)} \ge m_s \cdot X_{s,p} \qquad \text{for all p,s} \qquad [6]$$

$$\sum_{p,t,q} d_{s,q} \cdot Y_{p,t,q}\Big|_{k(s,p,t)} \ge t_s \cdot m_s \qquad \text{for all s.} \qquad [7]$$

The objective function [1] minimizes total costs across all planning units. Equation [2] accounts the total conservation costs in each country as product of habitat area times land rent summed over all planning units. Constraint [3] limits habitat areas in each planning unit to given endowments. Constraint [4] implements representation targets for all species but allows deviations if the number of planning units with occurrence data is below the representation target. Constraint [5] depicts minimum requirements of non-substitutable habitat types for relevant species and planning units. Constraint [6] forces the habitat area for the conservation of a particular species to be large enough to support viable populations of that species. The summation over habitat types depicts the choice between possible habitat alternatives. Constraint [7] ensures that the total population size equals at least the representation target times the minimum viable population size. This constraint is especially relevant for cases where the representation target is higher than the number of available planning units for conservation. For example, a representation target of ten viable populations with possible species occurrences in only nine planning units would under [7] require one or more planning units to establish enough habitat for more than one viable population.

2.2 Estimation of spatially explicit wetland distribution

The precision of empirical environmental assessments depends on the appropriateness of the underlying model equations but also on the quality of input data. For optimal wetland conservation planning, the spatial extent and distribution of wetlands and suitable restoration areas denote important input data. Hence, this study applies data from the empirical wetland distribution model SWEDI (Schleupner 2010).

The spatial wetland distribution model SWEDI is a geographic information system (GIS)-based model that relies on multiple spatial relationships of existing geographical data. It is developed as extraction tool to denote wetland allocations in Europe and covers 37 European countries at resolution of 1 km². The SWEDI model estimates the spatial distribution of European wetlands by distinguishing between existing functional wetlands and sites suitable for wetland restoration by considering recent land use options. The evaluation of existing wetlands relies on a cross-compilation of existing spatial datasets and extraction of spatial wetland information. The determination of potential wetland restoration sites is more complex. It involves the integration and interpretation of a variety of GIS datasets by assuming that there is a relationship between environmental gradients (Franklin 1995). Knowledge rules for each biogeographical region are defined based on analysis and observed correlation of independent variables such as climate, hydrology, soil, elevation and slope to analyse environment-wetland relationships. The information is extracted from spatial data, such as CORINE land cover (EEA 2000), European Soil Database (Joint Research Centre 2004), Bioclim (Busby 1991), Worldclim (Hijmans *et al.* 2005), Gtopo30 (USGS 1996), and Potential Natural Vegetation (Bohn and Neuhäusel 2003). In this manner regression parameters that vary across space are estimated with the advantage that they allow for regional differences in relationships (Miller *et* *al.* 2007). This is especially useful if concerning the broad European scale of the model. In combination with geographical data of potential natural vegetation, land use and land cover only those sites are selected by the model that fall within agricultural areas and forests. Urban and other sealed off areas and their direct vicinity are assumed to be unsuitable for wetland restoration. Furthermore, those sites that contain already existing conservation areas like salt marshes or valuable sparsely vegetated areas are also excluded from potential wetland restoration sites. The GIS tool ArcGIS9 is used for analysis.

As result SWEDI distinguishes three main wetland types that are further sub-divided into five wetland categories: wet forests (alluvial and swamp), wet grasslands (such as reeds and sedges; only one category), and peatlands (bogs and fens). Open waters (water courses and water bodies) are considered separately. However, a large part of the European wetland species that are included in the HABITAT model also need open water habitat. Spatial data on the extent of water courses and water bodies are derived from CORINE land cover (EEA 2000) and the Global Lakes and Wetlands Database (GLWD) (Lehner and Döll 2004).

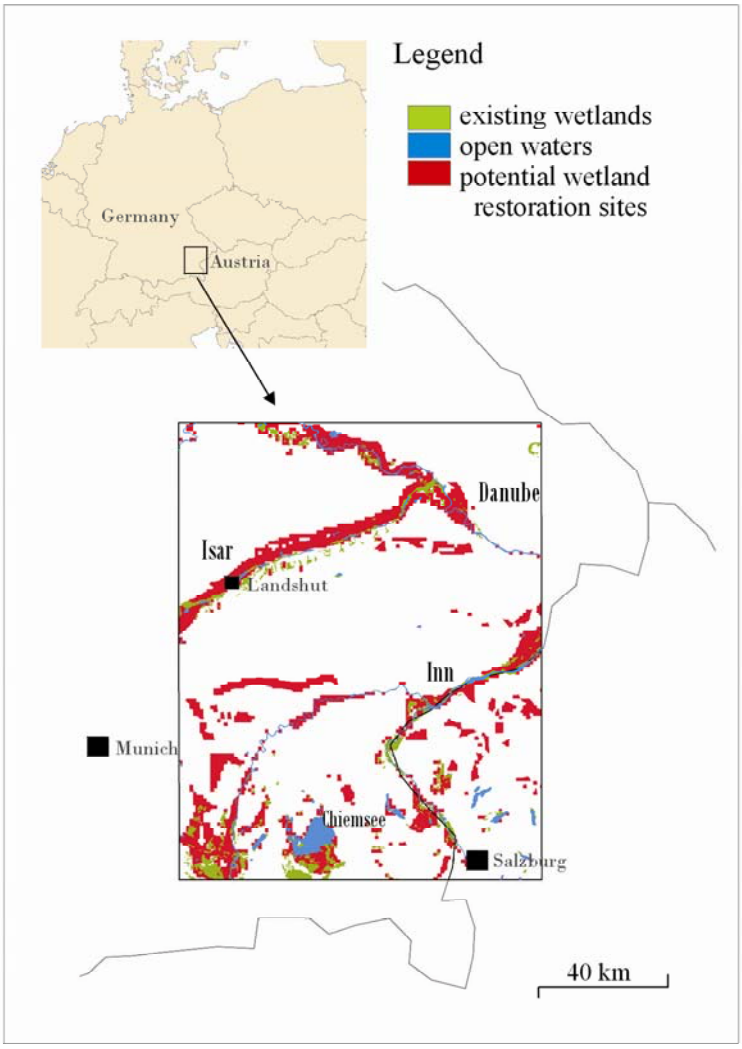

Figure III-2: Wetland areas from SWEDI for south-eastern Germany

Figure III-2 shows an extract of wetland areas from SWEDI for south-eastern Germany. The detailed map of the European wetland distribution and its potentials are described and illustrated in Schleupner (2009). The spatial planning units in the HABITAT model correspond to the resolution of the species occurrence data. We integrate the fine scale wetland data in terms of total areas of each wetland habitat type per planning unit.

2.3 *Spatially explicit data on land rents*

GEOSS data on land rents are estimated at Homogenous Response Unit (HRU) resolution covering the entire European continent. A HRU is a discrete characterization of land quality with pre-defined ranges on relatively stable attributes. Here, we use discrete classifications of altitude, slope and soil texture established through previous research (Skalsky *et al.*, 2008, based on previous works by Schmid *et al.*, 2006; Balkovič *et al.*, 2006; Stolbovoy *et al.*, 2007). HRUs are delineated on the assumption that within defined ranges of attributes, biophysical processes (e.g. plant growth, nutrient movement) respond similar to any set of exogenous impacts (e.g. rainfall, land management). Available data at HRU level include i) their spatial extent and ii) biomass yields and environmental impacts for major food and non-food cropping systems. The latter data are results from simulations with the Environmental Policy Integrated Climate model (EPIC, Izaurralde *et al.*, 2006; Williams, 1995). In addition, we use country specific land rents from the Global Trade Analysis Project model (GTAP, Lee *et al.* 2009). Based on these data, we approximate detailed GEOSS land rent data that are unique for each country and HRU.

We use the following notation: $u = \{1,\ldots,U)$ is the set of HRU; $c = \{1,\ldots,C\}$ is the set of countries. $s_{u,c}$ represents the share of a given HRU u within country c. $mr_{u,c}$ denotes the marginal revenue of land for HRU u in country c. v_c is a value parameter representing the difference between the weighted commodity price and all production costs except for the costs of land in country c. $i_{u,c}$ depicts the weighted average yield per hectare for HRU u and country c. mc_c represents the marginal costs of land in country c and $mc_{u,c}$ the marginal costs per HRU u in country c.

$$\sum_u s_{u,c} \cdot mr_{u,c} = \sum_u s_{u,c} \cdot i_{u,c} \cdot v_c = mc_c \qquad [8]$$

$$v_c = \frac{mc_c}{\sum_u s_{u,c} \cdot i_{u,c}} \qquad [9]$$

$$mc_{u,c} = i_{u,c} \cdot v_c \qquad [10]$$

Based on classic economic theory for competitive markets, equation [8] forces an identity between marginal revenues and marginal costs of land. While the marginal cost of land is given by its rental rate, the marginal revenue per hectare of land equals yield multiplied by a value parameter. The computation of the value parameter is shown in equation [9]. It depicts the difference between the weighted price of an agricultural or forestry commodity and its production costs. We assume that this value does not differ within a country. Finally, in equation [10] we compute HRU specific land rents by multiplying HRU specific yields by the value parameter.

Figure III-3 shows an extract of HRU specific land rents for south-eastern Germany. See supplementary material B for the land rents for all European countries. In the HABITAT model, HRU specific land rents in Euro per hectare are projected to all planning units. Since HABITAT does not distinguish different HRU within a planning unit, the land rents in each planning unit are area weighted averages over all contained HRU.

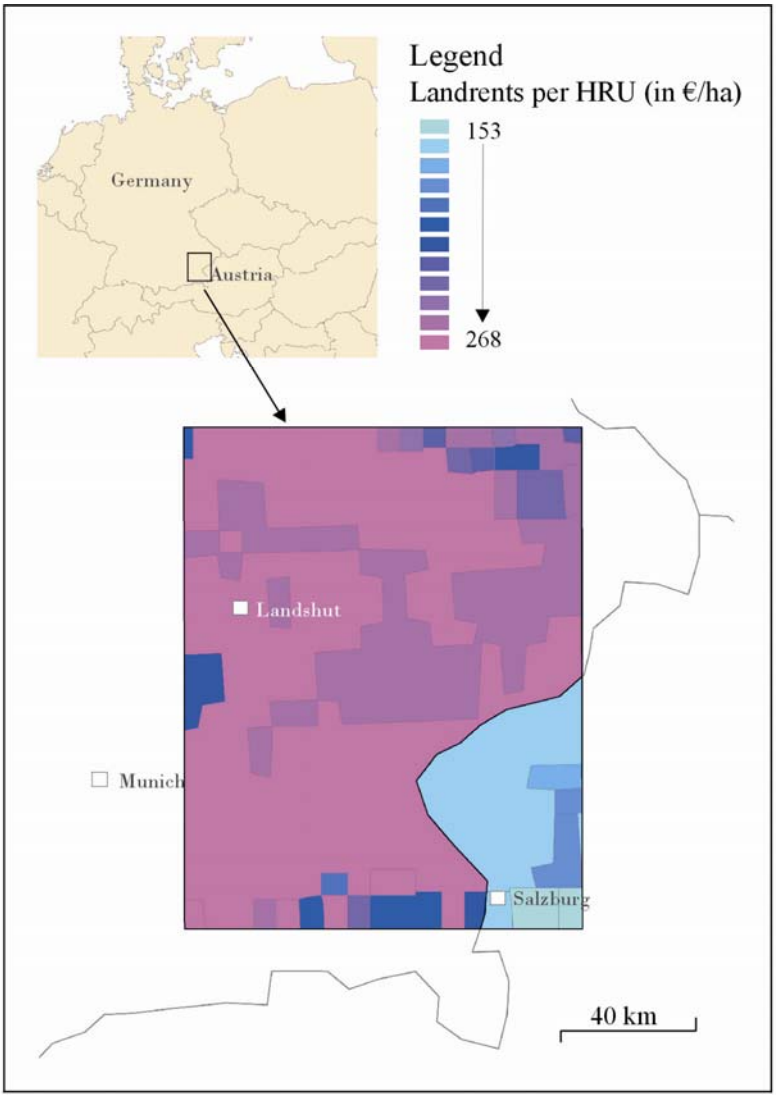

Figure III-3: Homogenous Response Unit specific land rents in south-eastern Germany

2.4 Model scenarios

To estimate the benefits to conservation planning from better resolved GEO data, we apply four scenarios of conservation planning. We compare high-resolution data on wetland habitat areas and land rents to frequently used low-resolution data.

In the first scenario, we use both coarse habitat and coarse land rent data. Under this setup, the total unsealed terrestrial area of each planning unit can be allocated to a species reserve provided that historical records of this species exist. Land rent data are taken from the GTAP model (Lee *et al.* 2009). These data differ only between countries, not within them. Hereafter, we refer to this scenario as **non-GEOSS scenario**. In the second scenario, we include fine scale wetland habitat data. However, the land rents remain uniform within each country. We label this setup as the **habitat-data scenario**. The third scenario examines the implementation of fine scale land rent data alone. In this **cost-data scenario**, habitat data are implemented at the coarse scale as done in the non-GEOSS scenario. Finally, the **GEOSS scenario** includes the fine scale datasets for both land rents and habitat areas.

3 Results

3.1 Costs of habitat protection and area requirements

Figures III-4 and III-5 show the estimated total conservation costs as function of the total required conservation area for each scenario and conservation targets 1 to 10. Annual costs for renting the land needed for habitat protection differ substantially between scenarios (see Figure III-4). The implementation of detailed habitat data alone incurs a mean increase of costs of habitat protection of 29.8% compared to the non-GEOSS scenario. On the other hand, integrating detailed cost data alone leads to an average cost reduction of 5.9%. Heterogeneous land rents within countries provide opportunities to select regions with below average rents and avoid regions with above average rents. Considering both factors simultaneously (GEOSS scenario), total land costs for habitat protection are on average 38.1% higher than those of the non-GEOSS scenario.

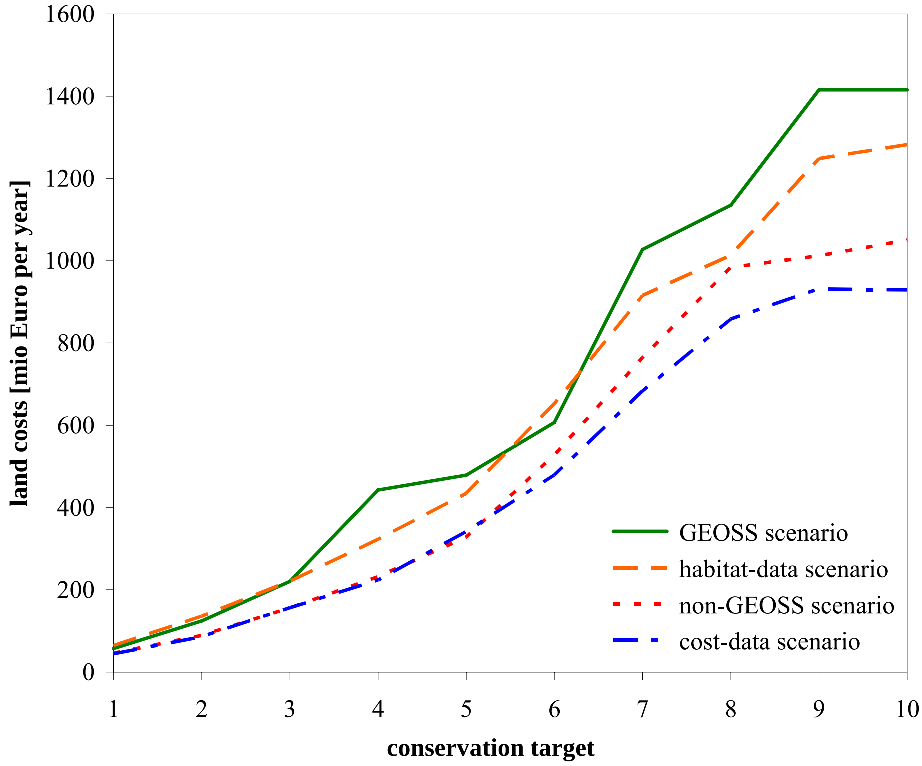

Figure III-4: Costs of habitat protection

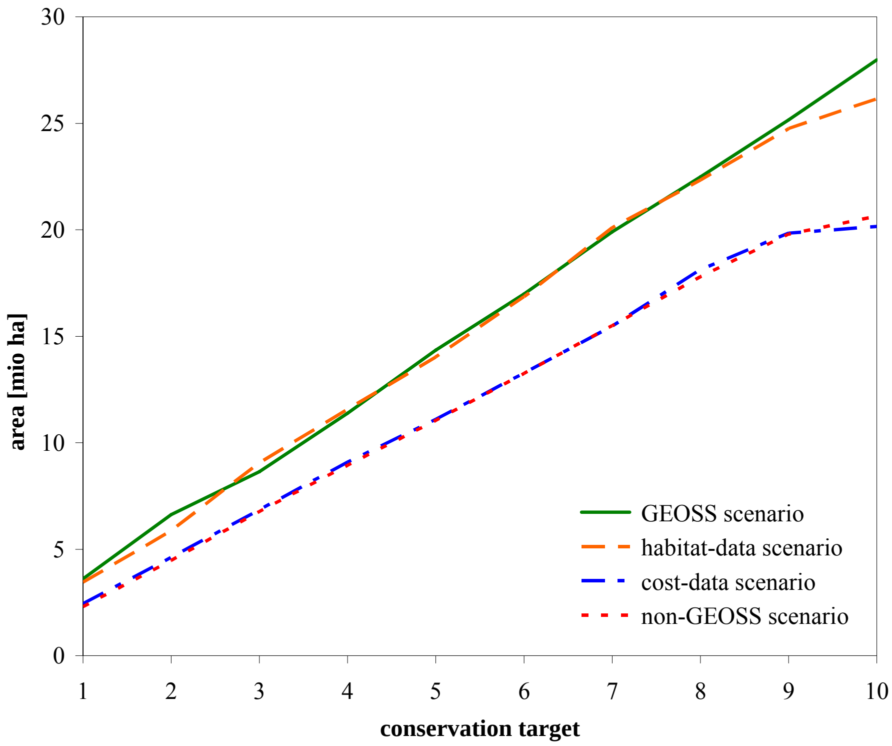

Figure III-5: Area requirements for conservation

The differences in conservation costs (Figure III-4) do not coincide with respective changes in area requirements (Figure III-5). Fine scale land rent data do not notably influence the extent of conservation areas compared to the baseline non-GEOSS scenario. The implementation of fine scale wetland habitat data implies higher overall area requirements in both scenarios containing these data. The area requirement of the habitat-data and the GEOSS scenario is on average about one third higher than the baseline scenario. This increase is due to the habitat type specifications that restrict reserve allocation to given endowments. With detailed area data, the model cannot exploit habitat synergies (one habitat simultaneously protecting multiple species) as much as with coarse data.

3.2 Benefits of fine scale data

The displayed cost estimates (Figure III-4) for scenarios with coarse scale data are biased and do not represent the true total costs. The bias arises from using i) incorrect data on land rents and and/or ii) incorrect habitat endowments. In addition, coarse data based solutions result in inefficient land allocations because the conservation planning model could place habitats to unsuitable or expensive locations. The analytical bias from using coarse scale data is illustrated in Figures III-6 and III-7, where we correct the results estimated under the non-GEOSS scenario to account for fine scale data. Particularly, we re-calculate conservation costs and target achievement but keep the size and locations of the conservation areas as determined under the setup with non-GEOSS data.

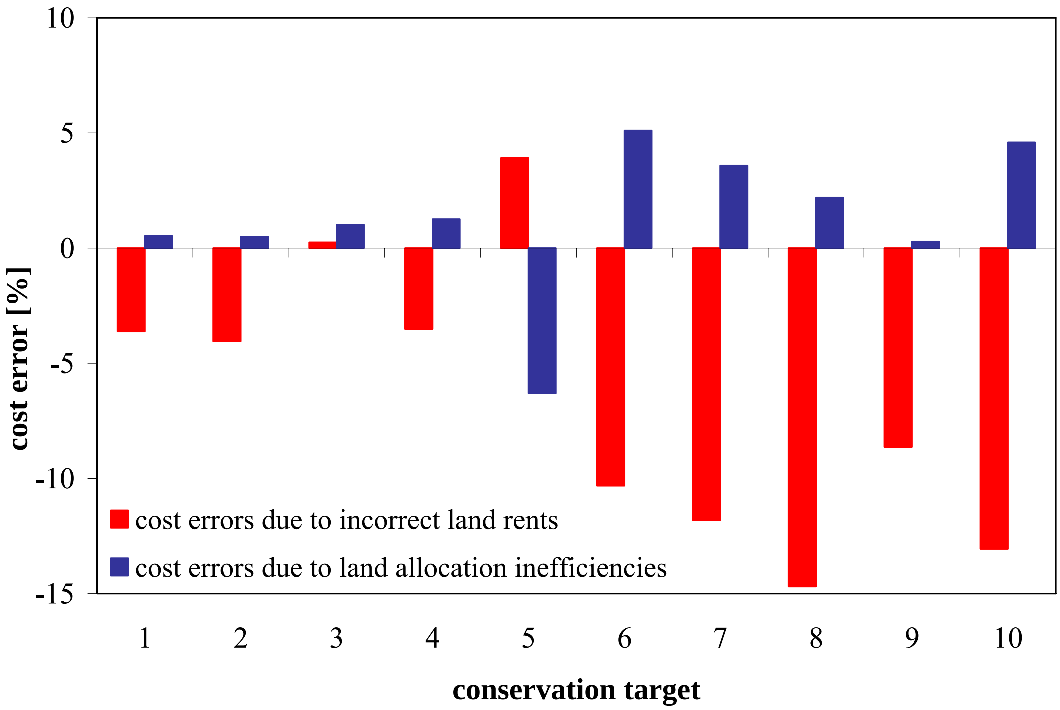

Figure III-6: Costs of non-GEOSS land rent data: errors in estimating conservation budgets. Shown are deviations compared to the cost-data scenario.

In the case of land rent data, costs of coarse scale data imply errors in the estimation of conservation budgets (Figure III-6). There are two types of errores. Misspecification of conservation costs due to incorrect land rents ranges from -14.7 to 3.9%. Cost errors due to land allocation inefficiencies range from -6.3 to +5.1%. In the case of habitat data, costs of coarse scale data imply losses in species coverage (Figure III-7). The species losses due to incorrect habitat data are substantial. Only 43 to 53 species out of 72 are covered according to the respective conservation target. Several species (3 to 19) are not covered at all throughout the targets. Benefits of GEO data for conservation planning thus encompass more accurate estimations on area requirements for conservation and costs of habitat protection.

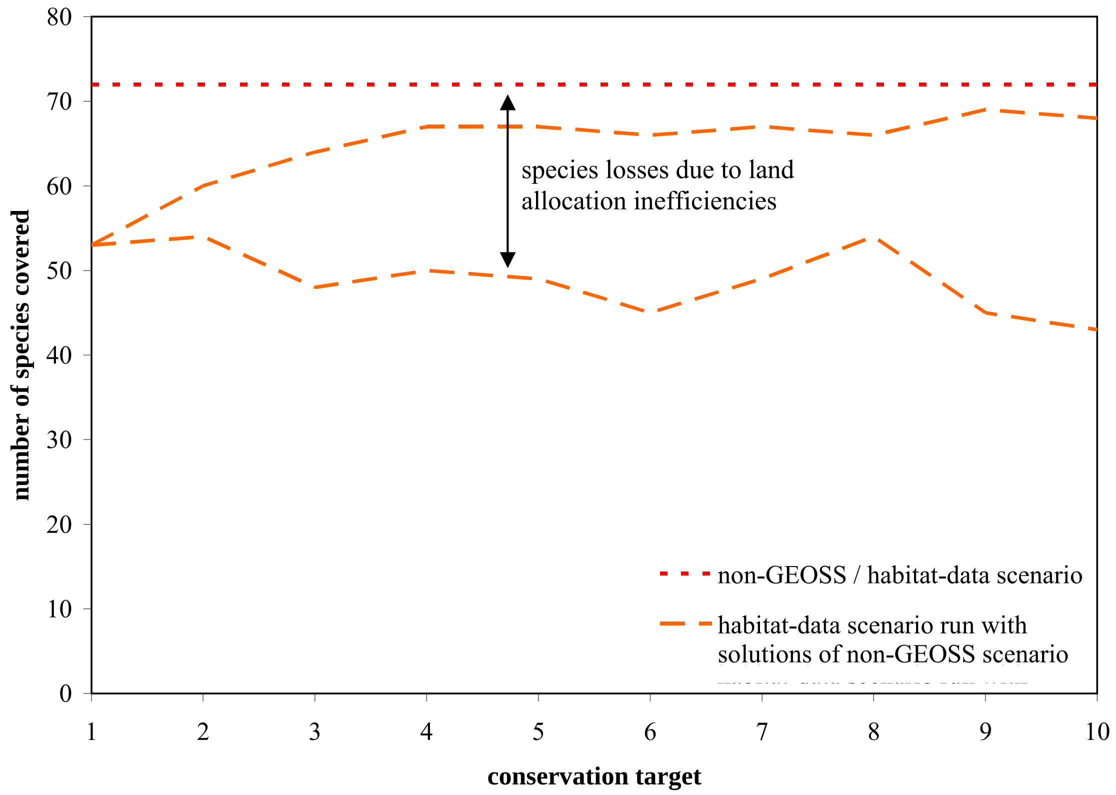

Figure III-7: Costs of non-GEOSS habitat data: losses of species coverage. The dotted line shows the number of species covered under the non-GEOSS and habitat-data scenario. The upper dashed line shows species covered under the habitat data scenario run with solutions of the non-GEOSS scenario. The lower dashed line shows species that are covered according to the respective target for the same model run.

3.3 Regional allocation of conservation areas

Implementation of fine scale data also affects regional reserve allocation between European countries (see Figure III-8). For the displayed conservation target, the required wetland area is in large part distributed between only 4 to 8 countries out of 37. The habitat type specifications force to spread the total required area across more countries. Implementation of detailed cost data does not have such a notable impact on the country scope but leads to changes of reserve shares between regions.

4 Discussions and conclusions

In order to effectively allocate scarce conservation funds, the use of conservation planning tools is inevitable (Margules and Pressey 2000, Possingham et al. 2000). However, the value of these tools depends on the availability and spatial resolution of required data. Our study shows that the employment of coarse scale data on habitats and land rents may lead to considerably cost-ineffective solutions in spatial conservation prioritization. This inefficiency is not easily detected because the reported total cost estimates are generally biased and therefore misleading. In our particular assessment, the conservation cost bias due to coarse scale cost data ranges from -14.7 to 3.9%. Furthermore, our non-GEOSS model version underestimates the area requirements for conservation by about 25%. Incorrect habitat data also imply substantial reduction in conservation target achievement.

We investigate and compare the effects of using different spatial resolutions for two datasets of a wetland biodiversity conservation planning tool. These include datasets on i) existing and potential wetland habitat areas and ii) land rents. A third spatial dataset is of utmost importance for these kinds of models, namely data about the distribution of biodiversity features (Margules and Pressey 2000). Although planning methods develop quickly, the availability of adequate biodiversity data limit their application (Prendergast *et al.* 1999, Pereira and Cooper 2006). Species occurrence data are used with one resolution only in our model as comprehensive data with a resolution higher than UTM50 are not available for Europe as a whole. Downscaling of the species data would be an option to work on smaller scales if a wide spatial scope such as the whole European continent is addressed (see Araujo *et al.* (2005) for an example on European plant and vertebrate species atlas data).

Please note that the implementation of comprehensive datasets in mathematical programming models may involve a substantial increase in computation time. This is especially true for integer programming. However, the commonly used branch&bound algorithms for this model type are well-suited to exploit parallel processor structures and computer grids.

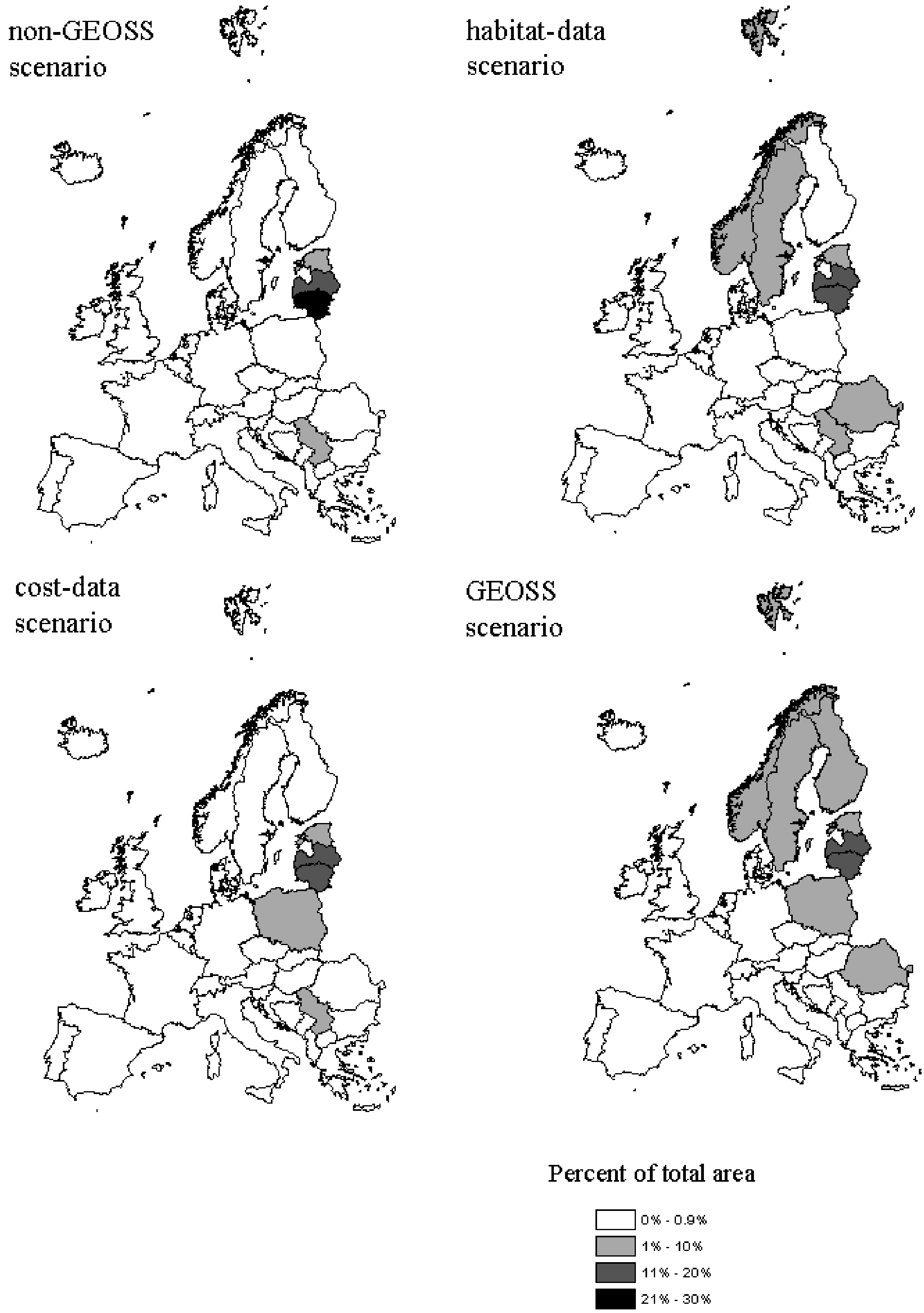

Figure III-8: Allocation of habitat area to European countries for conservation target 5

The knowledge of the extent and distribution of wetlands is important for a variety of applications. It is of utmost importance to provide accurate base data for the management and planning of conservation areas. This study applies SWEDI, an empirical distribution model to wetland ecosystems in European scale. SWEDI distinguishes three main wetland types for existing wetlands and for sites where wetlands could be restored or established. For the determination of existing wetland locations, several spatial datasets are jointly analysed. Potential wetland restoration sites are evaluated through geographic data analysis with utilization of rule-based statements as described in Schleupner (2010). The orientation towards physical parameters and the allowance of overlapping wetland types within the suitable restoration areas characterizes the SWEDI model. The accuracy of SWEDI model results is strongly restricted by the availability and quality of geographical data. For example, the soil information is generally poor and often misleading in regard to wetland functionality. Another uncertainty involves the current state of existing wetland ecosystems. SWEDI is not able to assess the naturalness of the site. Nevertheless, the validation with independent datasets of wetland biotopes such as RAMSAR sites proved high accuracy of the existing wetland sites in SWEDI and the area sizes are mainly reproduced within the uncertainty range (see Schleupner 2009). The detailed spatially explicit wetland classification of SWEDI allows connections to other habitat databases as well. The advantage of SWEDI is that the distribution modelling process is extended to a broad continental scale by keeping the spatial accuracy as high as possible. This is important because European wetlands are often fragmented ecosystems of small extent. Many wetlands are smaller than 1 km². Improvements in data quality and availability as well as simplifications in earth observation techniques make more detailed studies feasible. As a result through SWEDI the narrow stripes of alluvial forests or small isolated bogs may be better represented in broad-scale analyses of wetlands.

Homogenous response units arrange heterogeneous land attributes into discrete classes. Each combination of altitude, soil, and slope class is considered to be unique. However, within a certain class element, the response is considered to be homogenous. Thus, depending on the number of classes for each attribute, HRUs involve more or less approximation errors. For example, the first altitude class of our classification scheme ranges from the lowest level to 300 meters above sea level. All locations within this range are represented through the same weighted average altitude value. Furthermore, we use weighted, productivity based, marginal value differences as proxy for differences in land rental values between HRUs. In reality, other factors related to markets and local policies may influence local land rental values. Thus, our approach must be interpreted as first approximation until comprehensive land rent data for Europe are available.

Several simplifications of the HABITAT model should be noted. First, we include only the land opportunity costs from acquiring land for conservation. There are important additional costs, i.e. costs related to reserve establishment and maintenance (Naidoo *et al.* 2006). Second, we do not account for spatial reserve design criterions like connectivity or compactness and do not consider spatio-temporal aspects of persistence. We distinguish only between five coarse habitat classes with no quality differences. Another set of simplifications involves the conservation target approach. First, we employ the same conservation target for each species in our application. Main reason for this is the difficulty in consistently determining explicit targets for a range of individual species (Kerley *et al.*, 2003). Second, to account for persistence, reliable density data as well as proxies for MVP sizes are essential. However, density data vary substantially and are biased towards well-surveyed regions (Schwanghart *et al.* 2008). The employed absolute values of species' pairs, territories or families serve as proxies for viable populations. They are not assumed to represent real minimum viable populations, but are used as working targets due to the lack of better data. Similar proceeding can be found in Verboom *et al.* (2001) and Kerley (2003).

We conclude that conservation plans benefit from the integration of high resolution habitat area and land rent data. Better data lead to more accurate and more cost-efficient biodiversity protection plans with improved area allocations. For a given financial budget, the improved allocation of conservation areas also improves the species coverage. Our study results provide quantitative benefit estimates for one application of improved GEO data. We do not estimate the costs of obtaining improved land cover and land value information. Nevertheless, the benefits estimates could be used in broad cost-benefit assessments for existing or planned GEO systems. Fritz *et al.* (2008) develop a conceptual framework to assess benefits but also costs for a GEO information system.

Acknowledgements

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Supplementary Material A

Table III-A1: Ecological data on wetland species of European conservation concern

Shown are the 72 included species with their proxies for MVP sizes (adapted from Verboom *et al.* (2001)), density data, and habitat types. The genus *Discoglossus galganoi* includes *Discoglossus jeanneae*. For *Castor fiber*, the Estonian, Latvian, Lithuanian, Finnish and Swedish populations are excluded (according to 92/43/EEC). Regarding the densities for colonial birds, we differentiate nesting and foraging areas. The foraging area is set to 5 ha per reproductive unit (RU). Regarding the densities of the amphibian species, we assume 10 RU per hectare for solitary species and 20 RU per hectare for gregarious species. x stands for a required habitat type; / stands for an optional habitat type. The category open water is introduced for species that need some type of open water habitat. Area-demanding species (marked with an asterisk) are allowed to inhabit up to 50% non-wetland habitat in the habitat-data and GEOSS scenario.

			Required (x) and optional (/) habitat types					
Scientific name	**MVP [RU]**	**Maximum density [RU/ha]**	**Mire**	**Wet forest**	**Wet grassland**	**Water course**	**Water body**	**Open water**
Amphibians								
Alytes muletensis	200	20				x		
Bombina bombina	200	20			x		x	
Bombina variegata	200	20		/	/		x	
Chioglossa lusitanica	200	10				x		
Discoglossus galganoi	200	10					x	
Discoglossus montalentii	200	10				x		
Discoglossus sardus	200	10					x	
Pelobates fuscus insubricus	200	10					x	
Rana latastei	200	20		x			x	
Salamandrina terdigitata	200	10				x		
Triturus carnifex	200	10		/	/		x	
Triturus cristatus	200	10		/	/		x	
Triturus dobrogicus	200	10			/		x	
Triturus karelini	200	10					x	
Triturus montandoni	200	10		x	/		x	
Triturus vulgaris ampelensis	200	20		/	/		x	
Reptiles								
Elaphe quatuorlineata	120	2			/			
Emys orbicularis	120	15					x	
Mauremys caspica	120	9						x
Mauremys leprosa	120	9						x

			Required (x) and optional (/) habitat types					
Scientific name	**MVP [RU]**	**Maximum density [RU/ha]**	**Mire**	**Wet forest**	**Wet grassland**	**Water course**	**Water body**	**Open water**
Birds								
Acrocephalus paludicola	200	1.09			x			
Alcedo atthis	200	0.15						x
Anser erythropus	200	0.127		x				x
*Aquila chrysaetos**	120	0.0002	/		/			
*Aquila clanga**	120	0.000055	/	x	/	/	/	
Ardea purpurea purpurea	120	0.19			x			x
Ardeola ralloides	200	0.19			x		x	
Asio flammeus	200	0.1	/		/			
Aythya nyroca	200	1			x		x	
Botaurus stellaris stellaris	200	0.5			x			
Bucephala islandica	200	0.17						x
Chlidonias hybridus	200	0.19			/		x	
Chlidonias niger	200	0.19			x		x	
*Ciconia ciconia**	120	0.001415			x			*x*
*Ciconia nigra**	120	0.00018		x				x
Crex crex	200	0.19	/		x	/		
Fulica cristata	200	10			x		x	
Gavia arctica	120	0.006					x	
Gelochelidon nilotica	200	0.19			x	x		
Glareola pratincola	200	8			x		x	
*Grus grus**	120	0.00043	/	/	/		/	
Haliaeetus albicilla	120	0.01273		x				x
Histrionicus histrionicus	200	0.67				x		
Hoplopterus spinosus	200	0.3846			x			x
Ixobrychus minutus minutus	200	1.97			x			x
Marmaronetta angustirostris	200	0.19			x		x	
Milvus migrans	120	1.2733						x
Nycticorax nycticorax	200	0.19			x			x
Oxyura leucocephala	200	1.5					x	
*Pandion haliaetus**	120	0.0004		/			x	
Pelecanus crispus	120	0.19			/		x	
Pelecanus onocrotalus	120	0.19			/		x	
Phalacrocorax pygmaeus	200	0.19		/	/		x	
Philomachus pugnax	200	1	/		/			
Platalea leucorodia	120	0.19		/	x		x	
Plegadis falcinellus	200	0.19		/	x		x	
Porphyrio porphyrio	200	3.3			x		x	
Porzana parva parva	200	5			x		/	
Porzana porzana	200	0.333	/		/			
Porzana pusilla	200	3.5368			x			
Sterna albifrons	200	0.19				x	/	
Tadorna ferruginea	120	10					x	
Tringa glareola	200	0.12	x	/	/			

Scientific name	MVP [RU]	Maximum density [RU/ha]	Required (x) and optional (/) habitat types					
			Mire	Wet forest	Wet grassland	Water course	Water body	Open water
Mammals								
*Castor fiber**	120	0.002		x				x
Galemys pyrenaicus	200	13.89						x
*Lutra lutra**	120	0.00017						x
Microtus cabrerae	200	57.5			x			
Microtus oeconomus arenicola	200	65	/		/	/	/	
Microtus oeconomus mehelyi	200	65	/		/	/	/	
Mustela lutreola	200	0.083			/	x	/	
*Myotis capaccinii**	200	0.0042						x
*Myotis dasycneme**	200	0.0042					x	

Supplementary Material B

Table III-A2: Land rents for agricultural land: non-GEOSS and GEOSS data for 37 countries

The land rent data are based on the GTAP model (Lee *et al.* 2009). The GEOSS data additionally include HRU productivity data from the EPIC model (Izaurralde *et al*., 2006; Williams, 1995). For the countries indicated with an asterisk, HRU productivity data were not available. Data from neighboring countries or countries with similar HRU distribution were used for the calculation of GEOSS data (Andorra: data from Bosnia; Iceland: data from Norway; Liechtenstein: data from Austria).

	Land rent in Euro per hectare	
	non-GEOSS data	GEOSS data (range)
Albania	61.5	28.24-93.76
Andorra	39.03	27.40-32.89*
Austria	178.59	148.24-190.32
Belgium	180.96	137.53-218.31
Bosnia	39.03	20.71-49.82
Bulgaria	136.01	74.55-169.67
Croatia	83.97	67.38-140.09
Czech Republic	133.65	69.12-142.44
Denmark	286.22	285.52-287.52
Estonia	20.11	18.58-22.77
Finland	34.3	6.4-46.73
France	165.58	153.68-407.89
Germany	253.1	133.65-316.60
Greece	109.99	83.93-163.05
Hungary	99.35	8.85-101.53
Iceland	18.92	0.38-33.31*
Ireland	105.26	70.92-155.36
Italy	233	158.69-321.11
Latvia	10.64	10.23-20.34
Liechtenstein	18.92	18.38-21.57*
Lithuania	18.92	17.98-21.51
Luxembourg	180.96	157.99-377.94
Macedonia	39.03	32.79-84.40
Monaco	39.03	39.03
Montenegro	39.03	24.22-42.01
The Netherlands	379.66	346.43-578.17
Norway	18.92	0.38-33.31
Poland	108.81	78.27-165.49
Portugal	76.88	32.79-98.23
Romania	92.25	54.24-164.40
Serbia	39.03	24.22-42.01
Slovakia	55.59	5.01-60.09
Slovenia	52.04	22.32-59.93
Spain	83.97	11.96-165.24
Sweden	24.84	21.14-31.11
Switzerland	432.88	370.28-552.89
United Kingdom	160.85	112.77-350.10

IV

Gap analysis of European wetland species: priority regions for expanding the Natura 2000 network

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Abstract. Protected areas in the European Union under the Natura 2000 reserve system cover about 17 percent of the total land area. Systematic evaluations of the effectiveness of the current reserve system have been scarce and restricted to regional assessments. One reason for that may be the poor availability of comprehensive fine scale biodiversity data for the highly fragmented and densely human-populated European continent. We introduce a novel method applying principles from systematic conservation planning to conduct a detailed gap analysis using coarse scale species occurrence data. The applied mathematical programming model employs mixed integer programming techniques. We include fine scale wetland habitat data as well as species-specific proxies for minimum viable population sizes. First, we evaluate the performance of the current Natura 2000 system in covering endangered wetland vertebrate species. Results show that five area-demanding vertebrates are not covered by the current reserve system. Second, we identify potentials for expanding the network to move toward complete coverage for the considered species mostly in countries of North-Eastern Europe. 3.02 million hectares of additional reserve area at a cost of 106.56 million Euro per year would be required to ensure coverage of all considered species. Third, we present spatially explicit priority regions for expanding the reserve network cost-effectively.

Keywords: effectiveness of reserve systems, mathematical programming model, persistence, representation, population viability, systematic conservation planning

1 Introduction

Protected areas are the foundation for most national conservation policies. Accordingly, governments around the world have made commitments to establish systems of protected areas that conserve viable representations of terrestrial, freshwater, and marine ecosystems (IUCN 2003). Jenkins and Joppa (2009) estimate that 12.9% of the global terrestrial area is formally protected although only 5.8% is within strictly protected areas (IUCN categories I-IV). However, little is known of the extent to which these areas contribute to the goal of protecting biodiversity (Brooks et al. 2004; Rodrigues et al. 2004a). Many systems of protected areas are not representative of national biodiversity, as their selection is rather biased towards economically marginal landscapes (Pressey et al. 2002; Rouget et al. 2003).

Europe is one of the world's most densely human-populated continents and has a long and complex cultural history. The cornerstone of the nature and biodiversity policy in the European Union (EU) is Natura 2000. This EU-wide network of protected areas is regulated mainly by two directives: the 1979 Birds Directive and the 1992 Habitats Directive. The Birds Directive lists 193 bird species and subspecies of conservation concern for which the EU member states must designate Special Protection Areas (SPAs). The Habitats Directive identifies 231 natural habitats and 1180 plant and animal species of conservation concern for which Sites of Community Importance (SCIs) are required to be proposed. The SPAs and SCIs make up the Natura 2000 network, whose objective is to assure the long-term maintenance of Europe's endangered species and habitats at "favorable conservation status" (European Commission 2009a).

Without doubt, Natura 2000 is the most important initiative for biodiversity conservation in Europe (Gaston et al. 2008; Pullin et al. 2009). Weber and Christopherson (2002) call Natura 2000 the most ambitious supranational initiative for conservation that has ever been undertaken. It has been proposed as the main strategy to meet the target of significantly reducing or even halting biodiversity loss by 2010 (Balmford et al. 2005).

About 17% of the EU land area is currently designated as protected under Natura 2000 (European Commission 2009b). Despite these efforts, Europe will not achieve the target of halting the loss of biodiversity by the end of 2010 (European Environment Agency 2009; Butchart et al. 2010). Hoekstra et al. (2005) identify the vast majority of the European continent's terrestrial area as crisis ecoregions with extensive habitat degradation and limited habitat protection. The sufficiency index of the European Commission's Environment Directorate-General measures the degree to which European states have proposed sites that are considered sufficient to protect the habitats and species mentioned in Habitats Directive Annex I and II. It reveals considerable shortfalls in the progress of member states in designating protected areas (European Environment Agency 2009). Meanwhile, 40-85% of habitats and 40-

70% of species of European conservation concern have reached an unfavorable conservation status (European Environment Agency 2009). This trend also includes progressive declines in wetlands across Europe during the last decades (Jones and Hughes 1993).

The effectiveness of the Natura 2000 network in maintaining biodiversity has been assessed rarely (Rondinini and Pressey 2007; Maiorano et al. 2007; Gaston et al. 2008). Pullin et al. (2009) therefore demand a systematic evaluation of the effectiveness of the network of protected sites. Previous studies are limited to regional assessments. For example, Dimitrakipoulos et al. (2004) examine the Natura 2000 sites on the Greek island Crete. Their results show that the network is insufficient in representing regional plant biodiversity. Araujo et al. (2007) and Maiorano et al. (2007) find that the Natura 2000 network contributes notably to biodiversity protection in two EU regions, Italy and the Iberian Peninsula. Nevertheless, both studies conclude that the network needs to be strengthened and complemented by further protected areas. To our knowledge, the entire spatial entity of the EU with the complete species and habitats assemblage of the Natura 2000 related directives has so far not been assessed in such evaluation processes. Gaston et al. (2008) conclude that the assessment of the effectiveness of existing protected area systems in Europe is patchy and rather ill developed.

Gap analysis is a method to evaluate the performance of existing reserve systems and to identify focus areas for expansion. This planning approach has interrelated roots in two research areas. On the one hand, there is the Gap Analysis Program (GAP) in the U.S. (Scott et al. 1993), focusing on the comprehensiveness of existing protected area networks and the identification of gaps in coverage. Second, systematic conservation planning methods concentrate on the identification of priority areas for the expansion of reserve systems (Margules and Pressey 2000). Both elements are of importance in a useful gap analysis. We need to know the current status of biodiversity protection as well as to identify promising locations for additional protected areas to move toward complete coverage.

Numerous gap analyses at global (Rodrigues et al. 2004a, b; Jenkins and Joppa 2009) and regional scales (Fearnside and Ferraz 1995; Ramesh et al. 1997; Powell et al. 2000; Scott et al. 2001; Dietz and Czech 2005; Catullo et al. 2008; Nel et al. 2009) reveal that coverage of species and ecosystems by existing networks of protected areas is insufficient for the long-term maintenance of biodiversity.

The issue of gap analysis has reached attention in Europe only recently and many existing analyses are restricted to relatively small regions or individual countries. European forests are the focus of a gap analysis by Smith and Gillett (2000). Oldfield et al. (2004) show that most types of natural areas are underrepresented in the reserve system of England. Two studies by Maiorano et al. (2006, 2007) address terrestrial vertebrate species in the Italian protected areas. Araujo et al. (2007) find that the network of protected areas on the Iberian Peninsula needs to be strengthened and complemented by additional sites to adequately cover terrestrial plant and vertebrate species.

To evaluate the status of biodiversity and to determine how current conservation efforts can be improved, biodiversity monitoring is crucial (Balmford et al. 2005). Gaston et al. (2008) assume that the poor availability of fine scale biodiversity data hinders scientifically sound conservation planning in Europe. While species distribution data have been better mapped in Europe than in most other regions worldwide, there is a considerable gap between the spatial resolution of biodiversity data and that of habitat fragmentation (Araujo 2004; Gaston et al. 2008).

Gaston et al. (2008) argue that species atlas data are too coarse for most conservation planning exercises, principally because such areas are too large to serve as planning units in these assessments. Oldfield et al. (2004), for example, state that most protected areas in England are far smaller than the resolution of biodiversity data, making it difficult to know whether species recorded in a particular planning unit actually occur inside corresponding protected areas.

Nonetheless, there are several studies taking use of European biodiversity data despite their coarse resolution. Benayas and de la Montana (2003), for example, identify areas of importance for strengthening protected area systems in Spain with 50 x 50 km UTM species atlas data. A study by Araujo et al. (2007) employs UTM50 data to evaluate the effectiveness of Iberian protected areas in the conservation of terrestrial biodiversity.

In accordance with Margules and Pressey (2000) and Maiorano et al. (2006), we argue that conservation planning should not be delayed until improved biodiversity data are available. As biodiversity losses can be irreversible, delayed conservation actions may leave fewer options for the future. Here, we introduce a method to conduct a detailed gap analysis using coarse scale species occurrence data. Being aware of the limitations of our approach due to the data deficiencies, we discuss the possible implications on a potential widening of the Natura 2000 reserve system.

Our study aims to contribute to a systematic evaluation of the current Natura 2000 network. In view of the dramatic decline in wetlands across Europe during the last decades (Jones and Hughes 1993; European Commission 2007) and several recent studies highlighting notable gaps in protected area systems for freshwater ecosystems (Yip et al. 2004; Abellán et al. 2007; Sowa et al. 2007; Nel et al. 2009), our study focuses on freshwater wetland species and their habitats. Our analysis covers the entire European Union.

Similar to the poor availability of fine scale species distribution data, data on the spatial distribution of wetlands also do not exist on a sufficient spatial scale across Europe. To consider wetland habitats adequately in our analysis, we estimate high resolution wetland data from the integration of available data sources with the SWEDI model (Schleupner 2010).

Specifically, the aims of our study are: (1) to assess the performance of the existing Natura 2000 network in covering threatened vertebrate wetland species and their habitats with respect to representation and persistence, (2) to identify potentials for expanding the network cost-effectively, (3) to derive explicit maps delineating wetlands promising for an expansion of Natura 2000.

2 Methods

We summarize our steps used to perform a gap analysis into four stages (Figure IV-1). We (I) develop conservation targets to guide the assessment; (II) compile data on the planning region; (III) assess the performance of the current system of protected areas; and (IV) identify priority regions for expanding the system. These steps are similar to those proposed by Margules and Pressey (2000) for a systematic conservation planning assessment.

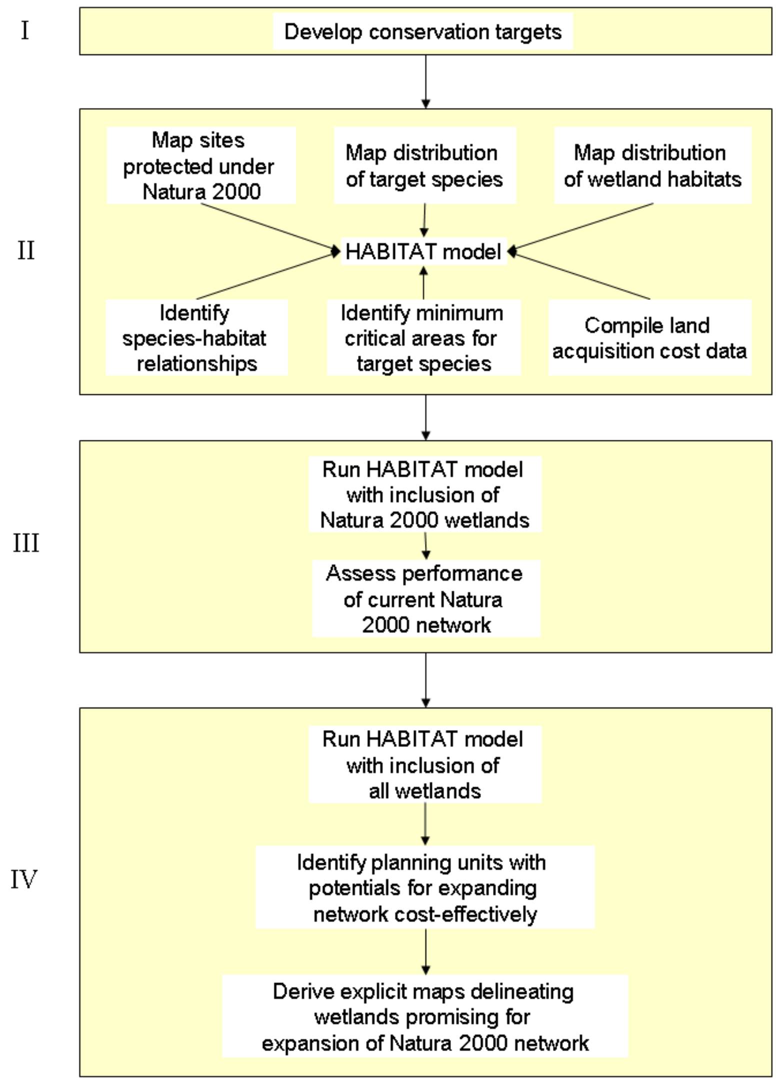

Figure IV-1: Flowchart of steps used to perform a gap analysis of European wetland species

2.1 Conservation targets

In Europe, as well as in most other world regions, explicit quantitative targets for the conservation of biodiversity are largely missing (Tear et al. 2005; Gaston et al. 2008). There are hence also no formal quantitative goals identified for the Birds and Habitats Directives and their species and habitats assemblage.

To be effective, a reserve network needs to represent all target species in protected areas that are large enough to ensure the long-term persistence of each species (Margules and Pressey 2000; Sarkar et al. 2006). According to our conservation target approach, a species is considered as covered by the existing reserve system, when (i) representation and (ii) persistence criterions are met simultaneously. A species is (i) represented when at least one occurrence is recorded inside Natura 2000 sites. We assume the (ii) persistence criterion to be fulfilled when two conditions are met. First, each representation corresponds to at least one viable population of that species. A population is considered viable when the allocated land area meets the minimum critical area (MCA), which is a species-specific measure based on density data and minimum viable population (MVP) sizes. Second, the land area that corresponds to the MCA of a species is allocated to habitat types required by that species. The concept of MCA is similarly applied in gap analyses of mammal species in Florida (Allen et al. 2001) and of primates in the Atlantic forest reserve system of Brazil (Pinto and Grelle 2009).

2.2 Data on the planning region

2.2.1 The study area and its existing Natura 2000 network

Our study area comprises the EU with 26 out of 27 member states. We exclude Cyprus and the Portuguese and Spanish islands in the Atlantic Ocean due to data deficiencies. The EU covers a terrestrial area of 4,324,782 km^2 of which approximately 40% is cultivated, while 4% are urban areas. About 500 million people inhabit the region, resulting in a population density of 116 inhabitants per km^2. The landscape is highly fragmented.

About 17% of the EU land area is protected under the Natura 2000 framework. As of November 2009, 22,419 SCIs with a total area of approximately 717,000 km^2 and 5,242 SPAs with a total area of approximately 575,000 km^2 have been submitted to the EU for approval. For SCIs, the national territory covered ranges from 6.8% in the United Kingdom to 31.4% in Slovenia. For SPAs, the percentage of national territory covered ranges from 2.9% in Ireland to 25.1% in Slovakia. About 90% of the reserves are smaller than 1,000 ha (European Commission 2009b). The spatial data on the Natura 2000 sites were provided by the European Commission, DG Environment (2008) (data on Austria and United Kingdom) and the European Environment Agency (2010) (updated data on other EU countries).

2.2.2 Target species

Freshwater wetland dependent species serve as surrogates for biodiversity. We include all 70 tetrapod wetland species listed in the appendices of the Birds and Habitats directive which encompass 16 amphibians, 4 reptiles, 41 breeding birds, and 9 mammals. Recorded occurrences from Gasc et al. (1997), Hagemeijer and Blair (1997), and Mitchell-Jones et al. (1999) identify their European distribution. The Universal Transverse Mercator (UTM) projection of the occurrence data results in grid squares of about 50 km edge length. The terrestrial parts of all 2237 grid cells encompassing the EU serve as planning units.

Species-specific MCAs are calculated from density data and MVP sizes. Species' density data were compiled through literature review; we use the maximum observed density. Proxies for MVP sizes are based on Verboom et al. (2001).

Data on habitat type requirements are taken from the literature as well. We distinguish five wetland habitat types including peatlands, wet forests, wet grassland, water courses, and water bodies. Furthermore, the type "open water" is applied to species that require either water courses or water bodies. We also distinguish required and optional habitat types. To enable the most area-demanding species to fulfill their area requirements, they are allowed to inhabit a certain share of non-wetland habitat. See Appendix A for the ecological data included for the 70 species.

2.2.3 Distribution of wetland habitats

Spatially explicit distributional data on existing functional wetlands and suitable wetland restoration areas are taken from the SWEDI model (Schleupner 2010). This empirical model comprises the most recent and comprehensive database on European freshwater wetland distribution. SWEDI distinguishes three main wetland types including peatlands, wet forests, and wet grasslands, at 1 km² resolution. Its GIS-based structure facilitates implementation into the HABITAT model with its UTM grid cell-based planning units via the spatial analyst functions of ArcGIS 9.3.

The knowledge of extent and distribution of open waters are also of importance for the performance of the HABITAT model. The required spatial data are extracted from CORINE (European Environment Agency 2000) and the Global Lakes and Wetlands Database (Lehner and Döll 2004). To put the current status of wetland protection in perspective, we differentiate three types within and outside Natura 2000 sites: a) recent existing wetland areas by wetland type, b) potential restoration areas by wetland type, c) open waters (sub-divided into water courses and water bodies). Table IV-1 shows the total areas of the above categories summed over the whole study region.

Due to scaling, uncertainties, and other deficiencies, these areas should only be considered as estimates rather than accurate observations. About 7% of all designated Natura 2000 sites are marked as wetlands in SWEDI, 4.3% are open waters, and another 12% might serve as suitable for wetland restoration. Overall, 31% of all recent wetland sites identified through SWEDI are protected under the Natura 2000 system.

Table IV-1: Wetland areas inside and outside Natura 2000 sites

	wetland category	inside Natura 2000 [in 1,000 ha]	outside Natura 2000 [in 1,000 ha]
recent wetland	peatlands	3,267.7	5,862.5
	wet forest	1,535.9	4,849.2
	wet grassland	246.3	523.8
potential wetland restoration area	peatlands	5,772.6[a]	41,495.3
	wet forest	3,617.3[a]	24,010.9
	wet grassland	4,408.1[a]	21,122.4
	total[b]	8,865.3	59,301.7
open water	water body	2,773.4	6,557.8
	water course	401.3	519.5

[a] Potential wetland restoration areas from the SWEDI model inside Natura 2000 sites are given for illustration purposes here, but are not included in the analysis.
[b] In the SWEDI model all three wetland types of the potential wetland restoration areas are allowed to overlap. The total area of potential sites is therefore not a summation of all wetland types.

2.2.4 Land cost data

Designating additional protected areas involves costs. These costs may include acquisition costs, management costs, transaction costs, and opportunity costs (Naidoo et al. 2006). Here, we address the acquisition and opportunity costs of land. These two cost types will usually equal if there is no market revenue from land after conservation and if there are no externalities involved in the alternative use (Bladt et al. 2009). Country-specific data on current agricultural land rents are taken from European land statistics (see Appendix B).

2.3 The HABITAT model

2.3.1 Planning units

HABITAT is a deterministic, spatially explicit model with many planning units of varying shape and size. Planning units are the spatial entities for which species occurrence data exist. We assume constant habitat suitability across all possible planning units. Parts of planning units necessary to fulfill conservation targets are selected as priority area for conservation. In case a species' MCA or habitat type requirement cannot be fulfilled within a single planning unit, the model selects further habitat in adjacent planning units. This approach differs from previous studies where either total planning units (e.g., Tognelli et al. 2008; Williams et al. 2005; Williams and Araujo 2002) or fractions of them (e.g., Bladt et al. 2009) are chosen. Rationale for our method is to overcome the problem of scale difference between the dimension of planning units and the available land area for conservation purposes. We have designed our model for planning units that are relatively large and/or located within densely human-populated regions. First, it is unlikely or even impossible to reserve such planning units entirely. Second, species' habitat size requirements will regularly not correspond to the dimension of a planning unit. Marianov et al. (2008) present a method to select reserves for species with differential habitat size needs exceeding planning units' areas. Our model also portrays area requirements which are smaller than a planning units' area. The total area selected as priority area for conservation in a planning unit includes the MCAs of all species protected in it.

2.3.2 Mathematical model structure

We use the following notation: $c = \{1,\ldots,C\}$ is the set of countries; $p = \{1,\ldots,P\}$ is the set of planning units; $t = \{1,\ldots,T\}$ is the set of habitat types; $q = \{1,\ldots,Q\}$ is the set of habitat qualities; $s = \{1,\ldots,S\}$ is the set of species. We employ several set mappings, which contain possible combinations between two or more individual sets. In particular, $u(s,t)$ identifies the mapping between species and habitat types and $k(s,p,t)$ the possible existence of species and habitats in each planning unit. The objective variable N represents the total number of conservation targets achieved by the included species. This variable is important for the first part of the gap analysis; the assessment of current protection levels of the Natura 2000 network. The objective variable O represents total opportunity costs. This variable is necessary for the second part of the gap analysis; the identification of priority regions for expanding the network. The non-negative variable array Z_c represents opportunity cost in country c. Another non-negative variable array $Y_{p,t,q}$ depicts the habitat area for planning unit p, habitat type t, and habitat quality q in hectares. $X_{p,s}$ is a binary variable array with $X_{p,s} = 1$ indicating planning unit p represents species s, and $X_{p,s} = 0$ otherwise. The model's exogenous data are given in small italic letters. $r_{c,p}$ denotes the annual land rent per hectare in country *c* and planning unit *p*. $a_{p,t,q}$ contains the maximum available area for planning unit *p*, habitat type *t* and habitat quality *q*. $d_{s,q}$ represents species- and habitat quality-specific population density data. m_s is a species-specific proxy for the minimum viable population size. $h_{t,s}$ determines non-substitutable habitat requirements for habitat type *t* and species *s*. t_s is the representation target for species *s*. v_s specifies possible deviations and equals the difference between the general representation target and exogenously calculated occurrence maxima.

Maximize $N = \sum_{p,s} X_{p,s}$ [1a]

Minimize $O = \sum_{c} Z_c$ [1b]

subject to:

$Z_c = \sum_{p \in c,t,q} Y_{p,t,q} \cdot r_{c,p}$ for all c [2]

$Y_{p,t,q} \le a_{p,t,q}$ for all p,t,q [3]

$\sum_{p} X_{p,s} \ge t_s - v_s$ for all s [4]

$\sum_{q} Y_{p,t,q} \ge h_{t,s} \cdot X_{s,p}$ for all p,t,s [5]

$\sum_{t,q} d_{s,q} \cdot Y_{p,t,q} \big|_{k(s,p,t) \wedge u(s,t)} \ge m_s \cdot X_{p,s}$ for all p,s [6]

$\sum_{p,t,q} d_{s,q} \cdot Y_{p,t,q} \big|_{k(s,p,t)} \ge t_s \cdot m_s$ for all s. [7]

The first objective function [1a] maximizes viable occurrences of species across all species and planning units. The second objective function [1b] minimizes total costs across all planning units. Note that in each simulation only one of these two objectives is active. Equation [2] calculates the total conservation costs in each country as product of habitat area times land price summed over all planning units. Constraint [3] limits habitat areas in each planning unit to given endowments. Constraint [4] implements representation targets for all species but allows deviations if the number of planning units with occurrence data is below the representation target. Constraint [5] depicts minimum requirements of non-substitutable habitat types for relevant species and planning units. Constraint [6] forces the habitat area for the conservation of a particular species to be large enough to support viable populations of that species. The summation over habitat types depicts the choice between possible habitat alternatives. Constraint [7] ensures that the total population size equals at least the representation target times the minimum viable population size. This constraint is especially relevant for cases where the representation target is higher than the number of available planning units for conservation. For example, a representation target of ten viable populations with possible species occurrences in only nine planning units would under [7] require one or more planning units to establish enough habitat for more than one viable population.

The problem is programmed in General Algebraic Modeling System (GAMS) and solved with a mixed integer programming algorithm from CPLEX version 12.1.

2.4 Assessment of current wetland biodiversity protection

The first part of this assessment estimates how much biodiversity is currently protected within the Natura 2000 network. In the model, we activate objective equation [1a] to maximize the number of distinct viable occurrences of species within the sites of the existing Natura 2000 network. The extent and habitat composition of the Natura 2000 sites are captured by the parameter $a_{p,t,q}$ which depicts the maximum available area for planning unit *p*, habitat type *t* and habitat quality *q*. To ensure that each species is covered at least once, the representation parameter t_s is set to 1.

The results of any gap analysis depend heavily on the criteria applied to distinguish between covered species and gap species (Rodrigues et al. 2004b). In this analysis, we apply three possible states depicting the coverage of a species inside a reserve system. We define a species as (i) fully covered if all recorded occurrences lie within protected areas and the corresponding habitat size equals for every occurrence at least the MCA for that species. If a species with several recorded occurrences fulfills the conservation target at least once, we consider it as (ii) covered, and otherwise to be a (iii) gap species.

2.5 Identification of priority regions for expanding the Natura 2000 network

As the existing Natura 2000 system does not fulfill the ambitious targets of national and international conservation objectives mentioned earlier, additional areas may be demanded to reduce or resolve the particular shortfalls. We address such demands in the second part of this assessment and determine the cost-minimizing locations of additional protected areas promising to move towards complete coverage.

In the model, we activate objective equation [1b] to minimize the total opportunity costs for a potential widening of the existing Natura 2000 network. The extent and habitat composition of the total unsealed land area inside and outside the Natura 2000 sites are depicted by the parameter $a_{p,t,q.}$ We set the lower bounds of the variable array $Y_{p,t,q}$ to the extent and habitat composition of the Natura 2000 sites. The representation parameter t_s is stepwise increased from 1 to 10 to force higher biodiversity benefits of an enhanced Natura 2000 system.

2.6 Delineation of potential sites for expansion

The identified priority regions for expanding the Natura 2000 network are downscaled with the SWOMP model (Schleupner 2009). SWOMP is based on spatial analyses using ArcGIS 9.2 as well as the analysis tools V-late and Hawths Analysis Tools (2006; Lang and Tiede 2003; Tiede 2005). Data on the distribution of existing and potential wetlands are taken from SWEDI. Through the ArcGIS Model Builder function and Python Scripting, the downscaling process is automated. In the model, the restoration variables are computed iteratively until the maximum wetland area defined by the expansion area per planning units and wetland type is reached. Figure IV-2 summarizes the model structure.

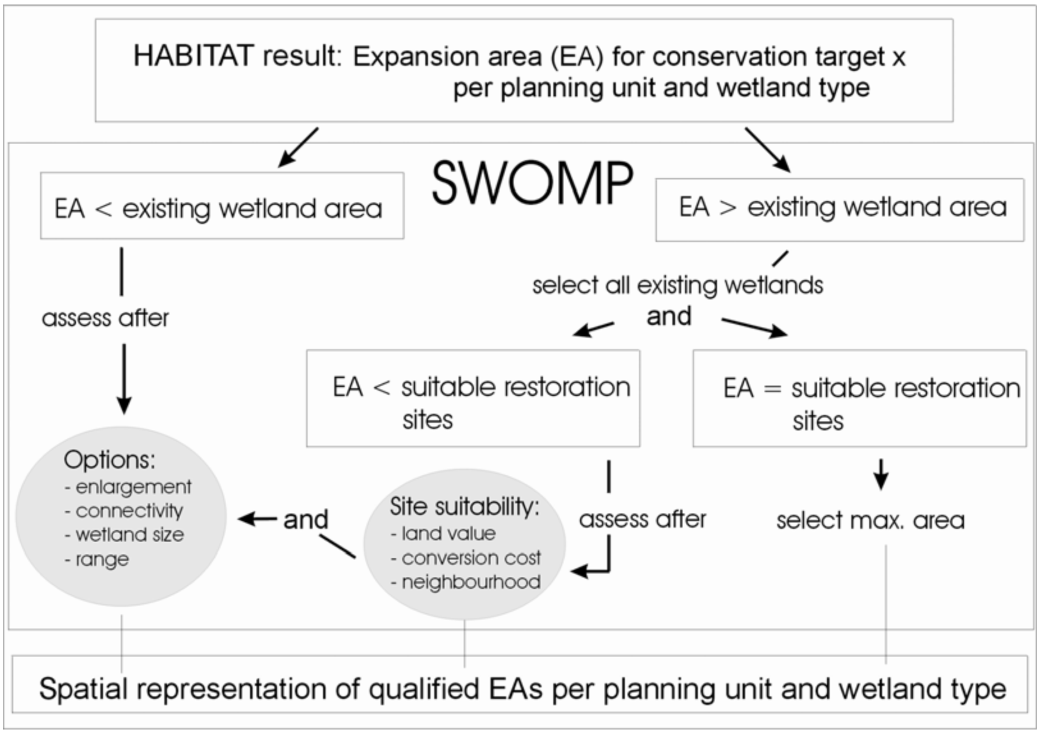

Figure IV-2: Overview of the downscaling model SWOMP

SWOMP gives preference to the protection of existing functional wetlands over restoration of degraded and conversion of other potential sites. The assessment of the most suitable sites relies on spatial criteria including enlargement (protected wetland sites might be enlarged by adjacent unprotected wetlands), connectivity (to build regional biotope complexes, evaluated by the proximity index after Gustafson and Parker (1992)), wetland size (determination of the desired minimum or maximum size of a wetland), and range (wetlands within a certain distance to other restored/existing wetlands or conservation areas of importance). The relative weight of individual criteria depends on the conservation objectives.

The determination of suitable wetland expansion sites also depends on their economic suitability. This suitability is based on three parameters including land value (opportunity costs of land to be converted into wetland), conversion cost (restoration success and costs valued after potential natural vegetation and land use), and neighborhood value (areas prioritized after area quality by using the hemeroby concept). The spatial-ecological criteria described above can be used optionally in addition to these three parameters to determine the most qualified sites within the allocated economic adequate areas. The result is a map showing the most promising sites for an expansion of the Natura 2000 network. For a detailed description of SWOMP see Schleupner (2009).

3 Results

3.1 Performance of current Natura 2000 network in covering wetland species

A total of 2194 planning units out of 2237 include at least a fraction of a Natura 2000 site. These planning units, which comprise an area of about 50 x 50 km or 250,000 ha, contain between 1 and 391 sites varying in size between <1 and 14,835 ha. This summary illustrates the high fragmentation of the Natura 2000 network on the densely human-populated European continent.

The first part of our gap analysis shows that only two species are (i) fully covered in the existing Natura 2000 system. All recorded occurrences of the Dutch root vole (*Microtus oeconomus arenicola*) and the Pannonian root vole (*Microtus oeconomus mehelyi*) lie within protected areas with their area requirements for viable populations fulfilled. Furthermore, we consider 61 other species as (ii) covered. According to our model, 21 species of this set are represented by hundred or more populations. We identify seven species as (iii) gap species, namely the spotted eagle (*Aquila clanga*), the golden eagle (*Aquila chrysaetos*), the black stork (*Ciconia nigra*), the osprey (*Pandion haliaetus*), the European otter (*Lutra lutra*), the Corsican painted frog (*Discoglossus montalentii*), and the Mallorcan midwife toad (*Alytes muletensis*). These seven species are represented inside several Natura 2000 sites, but their minimum area and/or habitat requirements are not met.

Given the coarse occurrence data, we need to assure that the species our model regards as covered by the Natura 2000 network are actually present in its protected areas. Therefore, we validate our results with the species lists of the Natura 2000 viewer (http://natura2000.eea.europa.eu/) and the EUNIS biodiversity database (http://eunis.eea.europa.eu/). First, although recorded as covered in our analysis, there are no recent records of the Lesser White-fronted Goose (*Anser erythropus*) within Natura 2000 sites in its breeding range. The Fennoscandian population of Lesser White-fronted Goose has declined rapidly since the middle of the 20th century and is facing an immediate risk of extinction (Jones et al. 2008; Tolvanen et al. 2009). There have been no confirmed breeding records of the original wild population after 1991 in Sweden (Tolvanen et al. 2009) and 1995 in Finland (Jones et al. 2008). As reintroduction initiatives are underway (Jones et al. 2008), we do not exclude the species from our analysis. Rather, we consider it as important to preserve the species' habitat which is according to our assumptions appropriate to sustain viable populations of that species. Second, for the two amphibian gap species, consultation of the databases revealed that they are well covered by several SCIs on the Spanish and French islands Mallorca and Corsica respectively. However, in our model, data inaccuracies pretended the nonexistence of their required habitats within the respective SCIs. Concerning the complex topic of data issues, we hereby refer to the discussion section. Figure IV-3 shows the number of fully covered, covered and gap wetland species after reclassifying the two amphibian species as covered (Corsican painted frog) and fully covered (Mallorcan midwife toad), respectively.

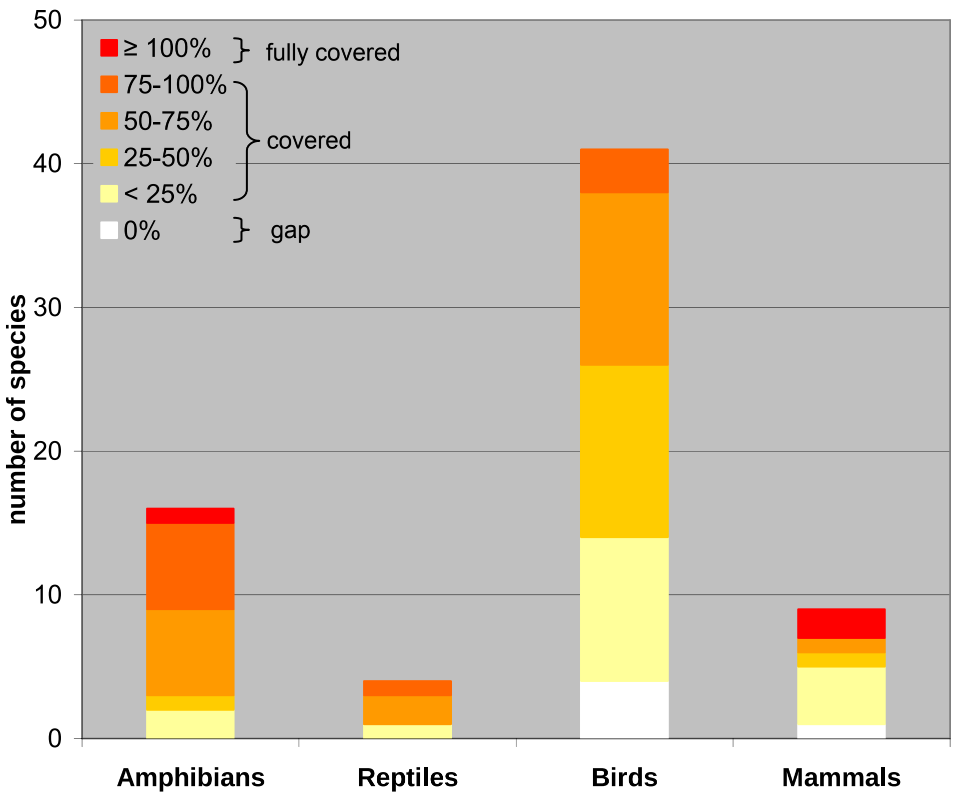

Figure IV-3: Number of fully covered, covered, and gap wetland species. Percentages indicate the degree of recorded occurrences covered by the Natura 2000 network

3.2 *Potentials for expanding the network cost-effectively*

To ensure that each considered wetland species is adequately covered with at least one viable population, the existing Natura 2000 network would require additional wetland habitats of 3.02 million hectare at a cost of 106.56 million Euro per year. The land area necessary for the cost-effective coverage of at least one viable population for each species is distributed mainly between the four EU countries Latvia (68.4%), Finland (19.4%), Estonia (12.0%), and Romania (0.2%).

One viable representation of a species in a reserve system, by definition, depicts only the absolute minimum to preserve this species over time within a relatively constant environment. Because of ecological and anthropogenic disturbances such as extreme weather events, epidemics, or certain economic activities, the minimum value will hardly guarantee long time survival. However, higher conservation targets will increases the overall cost. Figure IV-4 shows the area requirements and corresponding annual land costs of expansion for a range of additional conservation targets.

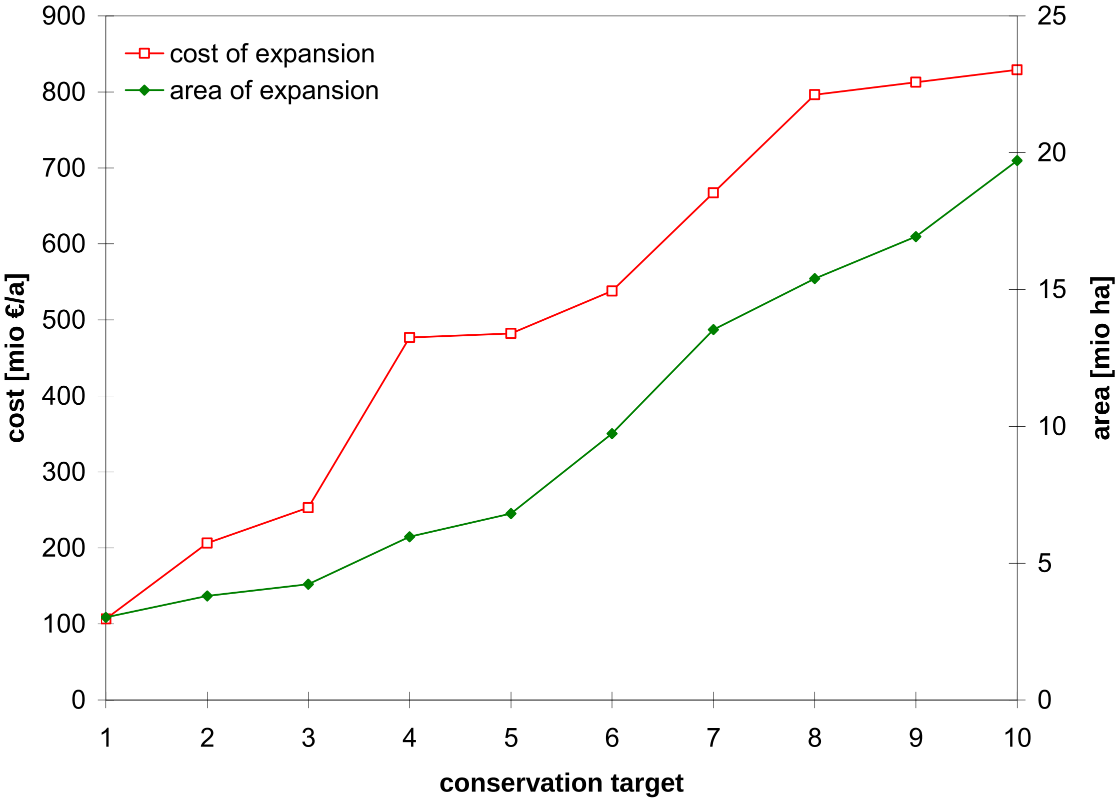

Figure IV-4: Additional cost and area requirements of an expanded Natura 2000 network for conservation targets 1 to 10

3.3 *Delineation of suitable sites for an expansion of the Natura 2000 network*

We apply SWOMP to downscale the estimates on expansion area per wetland type and planning unit from the HABITAT model. This process is illustrated below for the planning unit 2576 in Estonia. The unit is located between the Baltic Sea in the north and the Russian border to the east. The area contains two large Natura 2000 sites in the southern part, Muraka and Puhatu, which cover peatlands and wet forest complexes. For conservation target 1, the HABITAT model proposes to expand the Natura 2000 sites by 34,438 ha of wet forests and by 270 ha of water bodies. Figure IV-5 shows the selected planning unit with original and expanded wetland conservation areas.

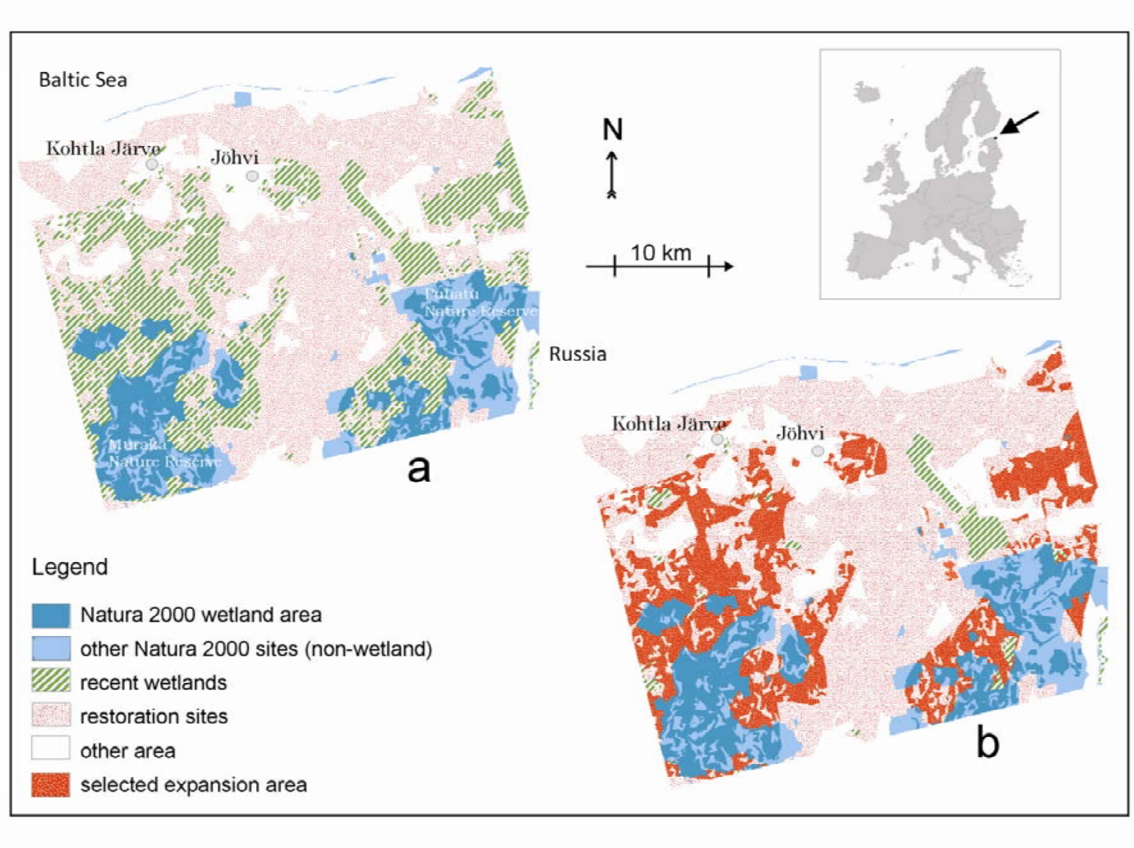

Figure IV-5: Downscaling example for planning unit 2576 in Estonia. a: current Natura 2000 and wetland distribution; b: SWOMP results based on HABITAT outcomes for conservation target 1

4 Discussions and conclusions

This study contributes to the complex issue of evaluating the efficiency and effectiveness of existing reserve systems. Two characteristics distinguish this analysis from previous ones. First, it takes use of coarse scale species occurrence data and still seeks to be spatially explicit. Second, we account for persistence by including species-specific habitat size requirements.

Our study provides an assessment of the effectiveness of the European Natura 2000 network of protected areas for the conservation of wetland dependant vertebrate species. We also identify species and regions that appear to be best candidates for expanding the existing reserve system cost-effectively in a densely human-populated landscape.

Not surprisingly, the existing scheme of protected areas does not represent all considered 70 tetrapod species adequately. Particularly, four wide-ranging wetland bird and one mammal species are not covered with a viable population. Explicit additional area requirement for gap species is part of the outcome of our model. However, results of any gap analysis depend critically on the applied conservation targets as well as on the quality of the underlying data (Scott et al. 1993; Maiorano et al. 2006). Changes in the dataset, especially in the population densities or the MVP sizes, could cause considerably different results.

We introduce a novel method to conduct a detailed gap analysis using coarse scale species occurrence data. Our planning units are about 50 x 50 km in size and reflect the scale of the available occurrence data. The common approach in conservation planning is to select planning units in its entity as priority area for conservation (Tognelli et al. 2008; Williams et al. 2005; Williams and Araujo 2002). Such procedure faces several problems, especially in Europe with its human-dominated landscape and high habitat fragmentation. First, from a policymaker's perspective, it will be unlikely or even impossible to reserve such planning units entirely. Second, many species' habitat size requirements do not correspond to the size of a relatively large planning unit. Third, suitable habitat areas for the maintenance of biodiversity may be scattered throughout a planning unit and not permit to meet all ecological criteria. There is a considerable scale difference between the dimension of planning units and the land area available for conservation (Araujo et al. 2004; Larsen and Rahbek 2003; Strange et al. 2006). See Cowling et al. (2003) for a discussion of scale-dependency on reserve selection. In contrast, our model selects as priority area for conservation only suitable parts of a planning unit. The identified habitat areas must meet the MCAs for all preserved species in each planning unit. To adequately represent the habitat composition in each planning unit, we integrate high resolution wetland habitat data.

This approach involves several limitations that need to be discussed. On the one hand, our analysis may overestimate species coverage inside reserves. First, the coarse data cause uncertainties. We do not know where exactly inside a UTM50 grid cell a species has been recorded and consequently cannot be sure that species match Natura 2000 reserves or proposed sites for expansion (see also Araujo 2004). To assure the species our model regards as covered by the Natura 2000 network are actually present in its protected areas, we validate our results with the Natura 2000 database (http://natura2000.eea.europa.eu/) and the EUNIS biodiversity database (http://eunis.eea.europa.eu/). In addition, we assume that suitable wetland habitats are sufficient indicators for wetland species occurrence. Thus, we compensate the deficiencies in species occurrence data by the inclusion of highly accurate habitat data. We consider a species protected when its required wetland habitat in a planning unit with recorded occurrences is protected. A second possibility for overestimation of species coverage is due to the relatively large planning units which prevent an explicit representation of each individual Natura 2000 site. The total Natura 2000 area in a planning unit may be built up from many small and scattered reserves which are not in close proximity to each other. Gaston et al. (2008), among others, raise concerns over the extent to which the European reserve systems can maintain biodiversity, given the small size of many protected areas. In our analysis of the Natura 2000 system, it may happen that although minimum area requirements of species are met, these areas are not made up by reserves that are connected in reality. This is especially critical for species with low dispersal abilities such as amphibians and reptiles. However, in the delineation of potential sites for expansion, we are able to address spatial reserve design criterions such as connectivity and compactness.

Our analysis may also underestimate species coverage inside reserves. First, our model may incorrectly classify some of the species as missing because of inaccurate global earth observation (GEO) data. The genus *Discoglossus montalentii,* for example, has occurrence records in five planning units. Within the boundaries of the corresponding Natura 2000 sites, not a single watercourse exists according to the employed GEO datasets CORINE (European Environment Agency 2000) and Global Lakes and Wetlands Database (Lehner and Döll 2004). The species fails to meet conservation targets and is recorded as a gap species. The same argumentation holds for the genus *Alytes muletensis* which is also recorded as gap species due to inaccurate habitat specifications. However, most amphibian species would need small ponds or ditches for breeding. At present, these habitats cannot be detected in satellite data. Second, in addition to statutory protected areas under the Natura 2000 framework, there are other European reserves which are not legally recognized but owned or managed by nongovernmental organizations or by private individuals (Gaston et al. 2008). These areas provide additional protection of wetland species of European conservation concern.

Given these limitations resulting from the use of coarse scale biodiversity data, this study emphasizes the need for European-wide biodiversity data on finer scales. Similar to other studies (Strange et al. 2006; Araujo et al. 2007; Gaston et al. 2008), we argue that the poor availability of these data remains a considerable constraint on conservation planning especially in Europe.

To estimate species-specific MCAs, we need to implement reliable data on population densities and MVP sizes. Observed population densities may vary substantially or be biased towards regions with high population densities (Schwanghart et al. 2008). We do not assume that the utilized values represent real MVPs or that defining explicit sizes for persistent populations is possible. Similar to other studies (Kautz and Cox 2001; Verboom et al. 2001; Kerley et al. 2003), we use these proxies as working targets given the lack of better data.

In agreement with other studies evaluating the effectiveness of the Natura 2000 system (Araujo et al. 2007; Maiorano et al. 2007), we find that the existing sites provide a limited degree of protection. To cover all species of European conservation concern adequately, the existing network needs to be expanded. To increase biodiversity benefits of the Natura 2000 network in a cost-effective manner, the expansion of protected areas should be coordinated across national borders. The collection and disclosure of highly resolved data plays an important role for systematic conservation planning. Further research is required to evaluate the significance of Natura 2000 sites for biodiversity features we did not include in our study, for example invertebrates, plant species, and vegetation communities.

Finally, we would like to note that we do not seek to undermine the significance of Natura 2000 and the many efforts leading to its existence. We rather intend to highlight the problem of population viability in a reserve system built in a highly fragmented and human-dominated landscape. As Maiorano et al. (2006, 2007) suggest, a chance would be to manage the matrix around Natura 2000 sites as a functional part of the reserve system. Where an expansion is not feasible in the near future, the priority regions identified in our study may serve as starting points for such a matrix management (see Prevedello and Vieira (2010) for a review).

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Appendix A

Table IV-A1: Wetland species of European conservation concern

Shown are the 70 included species with their proxies for MVP sizes (adapted from Verboom et al. (2001)), density data, and habitat type requirements.

Scientific name	MVP (RU)	Maximum density[c] (RU/ha)	Required (x) and optional (/) habitat types					
			Peatlands	Wet forest	Wet grassland	Water course	Water body	Open water[d]
Amphibians								
Alytes muletensis	200	20				x		
Bombina bombina	200	20			x		x	
Bombina variegata	200	20		/	/		x	
Chioglossa lusitanica	200	10				x		
Discoglossus galganoi[a]	200	10					x	
Discoglossus montalentii	200	10				x		
Discoglossus sardus	200	10					x	
Pelobates fuscus insubricus	200	10					x	
Rana latastei	200	20		x			x	
Salamandrina terdigitata	200	10				x		
Triturus carnifex	200	10		/	/		x	
Triturus cristatus	200	10		/	/		x	
Triturus dobrogicus	200	10			/		x	
Triturus karelini	200	10					x	
Triturus montandoni	200	10		x	/		x	
Triturus vulgaris ampelensis	200	20		/	/		x	
Reptiles								
Elaphe quatuorlineata	120	2			/			
Emys orbicularis	120	15					x	
Mauremys caspica	120	9						x
Mauremys leprosa	120	9						x
Birds								
Acrocephalus paludicola	200	1.09			x			
Alcedo atthis	200	0.15						x
Anser erythropus	200	0.127		x				x
*Aquila chrysaetos**	120	0.0002	/		/			
*Aquila clanga**	120	0.000055	/	x	/	/	/	
Ardea purpurea purpurea	120	0.19			x			x
Ardeola ralloides	200	0.19			x		x	
Asio flammeus	200	0.1	/		/			
Aythya nyroca	200	1			x		x	
Botaurus stellaris stellaris	200	0.5			x			
Chlidonias hybridus	200	0.19			/		x	
Chlidonias niger	200	0.19			x		x	
*Ciconia ciconia**	120	0.001415			x			*x*
*Ciconia nigra**	120	0.00018		x				x
Crex crex	200	0.19	/		x	/		
Fulica cristata	200	10			x		x	

Scientific name	MVP (RU)	Maximum density[c] (RU/ha)	Required (x) and optional (/) habitat types					
			Peatlands	Wet forest	Wet grassland	Water course	Water body	Open water[d]
Gavia arctica	120	0.006					x	
Gelochelidon nilotica	200	0.19			x	x		
Glareola pratincola	200	8			x		x	
*Grus grus**	120	0.00043	/	/	/		/	
Haliaeetus albicilla	120	0.01273		x				x
Hoplopterus spinosus	200	0.3846			x			x
Ixobrychus minutus minutus	200	1.97			x			x
Marmaronetta angustirostris	200	0.19			x		x	
Milvus migrans	120	1.2733						x
Nycticorax nycticorax	200	0.19			x			x
Oxyura leucocephala	200	1.5					x	
*Pandion haliaetus**	120	0.0004		/			x	
Pelecanus crispus	120	0.19			/		x	
Pelecanus onocrotalus	120	0.19			/		x	
Phalacrocorax pygmaeus	200	0.19		/	/		x	
Philomachus pugnax	200	1	/		/			
Platalea leucorodia	120	0.19		/	x		x	
Plegadis falcinellus	200	0.19		/	x		x	
Porphyrio porphyrio	200	3.3			x		x	
Porzana parva parva	200	5			x		/	
Porzana porzana	200	0.333	/		/			
Porzana pusilla	200	3.5368			x			
Sterna albifrons	200	0.19				x	/	
Tadorna ferruginea	120	10					x	
Tringa glareola	200	0.12	x	/	/			
Mammals								
Castor fiber[b],*	120	0.002		x				x
Galemys pyrenaicus	200	13.89						x
*Lutra lutra**	120	0.00017						x
Microtus cabrerae	200	57.5			x			
Microtus oeconomus arenicola	200	65	/		/	/	/	
Microtus oeconomus mehelyi	200	65	/		/	/	/	
Mustela lutreola	200	0.083			/	x	/	
*Myotis capaccinii**	200	0.0042						x
*Myotis dasycneme**	200	0.0042					x	

[a] The genus *Discoglossus galganoi* includes *Discoglossus jeanneae*.

[b] For *Castor fiber*, the Estonian, Latvian, Lithuanian, Finnish, and Swedish populations are excluded (according to 92/43/EEC).

[c] Regarding the densities for colonial birds, we differentiate nesting and foraging areas. The foraging area is set to 5 ha per reproductive unit (RU). Regarding the densities of the amphibian species, we assume 10 RU per hectare for solitary species and 20 RU per hectare for gregarious species.

[d] The category open water is introduced for species that need some type of open water habitat.

[*] Wide-ranging species are indicated with an asterisk.

Appendix B

Table IV-A2: Agricultural land rents for European countries

	Rent for agricultural land [€/ha*a][a]
Austria	244.53
Belgium	151.76
Bulgaria	70.19
Czech Republic	23.17
Denmark	315.00
Estonia	15.76
Finland	152.08
France	109.35
Germany	156.32
Greece	402.98
Hungary	54.56
Ireland	212.76
Italy	248.42
Latvia	8.34
Lithuania	17.14
Luxembourg	150.38
Malta	115.44
The Netherlands	396.01
Poland	68.08
Portugal	158.51
Romania	8.58
Slovakia	13.33
Slovenia	86.21
Spain	145.40
Sweden	98.12
United Kingdom	190.34

[a] data derived from Eurostat (averaged data from 1985 to 2006 for Austria, Belgium, Bulgaria, Denmark, Finland, France, Germany, Greece, Hungary, Ireland, Lithuania, Luxembourg, Malta, The Netherlands, Poland, Romania, Slovakia, Spain, Sweden, United Kingdom) and Farm Accountancy Data Network (FADN) (data from 2004 for Czech Republic, Estonia, Italy, Latvia, Portugal, Slovenia)

Overall Conclusions and Outlook

1 Conclusions

One aim of this thesis was to investigate ways to facilitate and strengthen the application of systematic conservation planning methods to European conservation problems. Data deficiencies hamper the application of common planning tools that were originally designed for other world regions. A second aim of this interdisciplinary thesis was to foster a better understanding and correct implementation of economic concepts in conservation planning applications. Given scarce monetary resources for conservation activities and high competition for land, the optimal allocation of conservation funds is inevitable to achieve conservation objectives. As conservation planning falls usually in the realm of biologists, economic considerations are often neglected.

The tool to address the thesis' objectives was the HABITAT model; a deterministic, spatially explicit mathematical programming model constructed in the General Algebraic Modeling System (GAMS). The model was applied to several spatial scopes, corresponding species assemblages, and objectives (see Table 1). Advancements in the field of systematic conservation planning through this thesis can be best outlined by distinguishing between general novel aspects of the HABITAT model and specific aspects studied in the individual chapters of the thesis.

First, the HABITAT model applies the conservation target approach as an advancement of the commonly used representation target. Systematic conservation planning relies fundamentally on two principles; representation and persistence of biodiversity features (Margules and Pressey, 2000; Sarkar et al., 2006). Previous studies argue that the emphasis in conservation planning assessments lies on representation whereas persistence is often neglected or inadequately addressed (Cabeza and Moilanen, 2001; Haight and Travis, 2008). In the HABITAT model, the same value is given to both factors by integrating them in the so-called conservation target. A species meets a conservation target only when (i) it is represented according to the target inside a reserve system and simultaneously (ii) its minimum area requirements for the respective viable populations are fulfilled. When addressing species as surrogates for biodiversity, as often the case in conservation planning, this method allows to directly account for persistence considerations.

Table 1: HABITAT model scope and configuration for studies of chapters I-IV

	Chapter I **Multiple-species conservation planning**	**Chapter II** **Integrating land market feedbacks**	**Chapter III** **Benefits of global earth observation**	**Chapter IV** **Gap analysis of European wetland species**
Model scope				
included countries	EU 27	EU 25	Europe	EU 27
number of planning units	2235	1996	2725	2237
number of species	70	69	72	70
amphibians	16	15	16	16
reptiles	4	4	4	4
birds	41	41	43	41
mammals	9	9	9	9
habitat areas	not specified	spatially explicit	not specified/ spatially explicit	spatially explicit
cost data	not specified	country-average	country-average/ spatially explicit	country-average
Model configuration				
objective	area minimization	cost minimization/ endogenous cost minimization	cost minimization	representation maximization/ cost minimization
optimization mode	sequential/joint	joint	joint	joint
target level	1-20	1-25	1-10	1-10

Second, closely connected with the conservation target approach is the endogenous representation of reserve sizes in the HABITAT model. Common reserve selection tools apply the basic formulation of the set-covering problem from operations research where planning units are only selectable in their entirety as priority areas for conservation. As there is a considerable gap between the resolution of European-wide species occurrence data and the land area available for conservation purposes in Europe, the application of these tools is limited. In the HABITAT model, the set-covering problem is extended. A planning unit is not necessarily selected in its entirety as conservation area, but only those fractions of a planning unit which are (i) necessary to fulfill the respective conservation target and (ii) theoretically available for reservation under the given land use pattern. Marianov et al. (2008) recently proposed a method to select reserves for species with differential habitat size needs exceeding planning units' areas. Our approach goes beyond that by also considering the fact that species' area requirements may be smaller than a planning units' area. The total area selected as priority area for conservation in a planning unit includes the minimum critical areas of all species protected in it. This procedure allows easy implementation of planning units with varying sizes. Thus, the HABITAT model does not only address persistence criterions directly, but also regardless of the planning unit's size. In combination with downscaling tools such as the SWOMP model presented in Chapter IV, we are finally able to present spatially explicit results on a resolution of 1km² despite the coarse biodiversity input data.

In addition to these general characteristics of the HABITAT model, several achievements of the individual studies need to be noted. The first study investigates the area efficiency of different degrees of geopolitical coordination in conservation planning. It compares five scenarios, including taxonomic, political, and biogeographical coordination of planning. Though the results are intuitive, this study for the first time illustrates and quantifies the considerable potential for area savings through meaningful cooperation beyond the borders of taxonomic groups, countries, or biogeographical regions.

The second study introduces a method to represent land acquisition costs endogenously in reserve selection. This highly interdisciplinary paper integrates concepts from ecological and economic theory. The underlying equations of the HABITAT model are modified to account for a dynamic representation of marginal costs, thereby explicitly integrating land market feedbacks. Results show that land markets may influence conservation efforts, because setting aside land for conservation itself changes land costs. The study confirms that ignoring these land rent adjustments can lead to highly cost-ineffective solutions in reserve selection.

The third study investigates benefits of improved land cover and land value information for conservation planning. Results show that the accuracy of conservation plans improves considerably with higher resolution habitat data and spatially explicit land rent data. In this paper, data on habitat distribution and land rents are obtained from various source datasets. Spatially explicit wetland data are taken from the geographical Spatial Wetland Distribution model (SWEDI, Schleupner, 2010). Economic theory is applied to derive spatially explicit land rents from base data taken from the biophysical Environmental Policy Integrated Climate model (EPIC, Izaurralde et al., 2006; Williams, 1995) and the economic Global Trade Analysis Project model (GTAP, Lee et al., 2009). The study shows how data deficiencies may be overcome by integrating available datasets from different models and sources.

The fourth study evaluates the performance of the current Natura 2000 system in covering endangered wetland vertebrate species and identifies potentials for expanding the network. Determining the effectiveness of protected area systems in covering biodiversity features is a core issue in systematic conservation planning (Margules and Sarkar, 2007). We conduct a detailed European-scale gap analysis despite the given data deficiencies. The delineation of spatially explicit priority areas for an expansion of the Natura 2000 network is processed by the combination of the reserve selection model HABITAT and the downscaling tool SWOMP (Schleupner, 2009).

However, the concept of reserve selection under the systematic conservation planning philosophy involves several simplifications and limitations. First, precise planning and efficient land allocation is only possible when conservation planning tools are used with adequate and reliable data. Given its complexity and different levels of organization, it is impossible to sample the full range of biodiversity. Relatively well-known taxonomic groups such as birds, butterflies, vertebrate species in general, and plant communities often serve as surrogates for biodiversity. The level of support for surrogates has been variable in the literature (Beger et al., 2003; Faith et al., 2004; Reyers and van Jaarsveld, 2000). Nevertheless, given the urgency associated with many conservation decisions, surrogates are essentially required to perform conservation planning assessments.

Consequently, the species atlas data serve as a critical model input dataset. Three atlases provide information on the European distribution of vertebrate species, namely the Atlas of Amphibians and Reptiles in Europe (Gasc et al., 1997), the EBCC Atlas of European Breeding Birds (Hagemeijer and Blair, 1997), and The Atlas of European Mammals (Mitchell-Jones et al., 1999). Solely the breeding bird atlas provides - for parts of its dataset - presence/absence data; all other species records are presence-only data. Apart from a potential bias in sampling effort, these data also do not account for recent declines in species abundance and distribution (i.e. the case of the Lesser White-Fronted Goose outlined in Chapter IV). Thus, reliability of results would substantially benefit from repeated and up-to-date monitoring activities on a continent-wide scale.

Second, limitations of the model itself need to be discussed. HABITAT is a pure reserve selection model. One shortcoming of these kinds of models is that they do not consider the spatial distribution of selected sites (Moilanen et al., 2009). The models ignore reserve proximity, connectivity, and shape. Consequently, solutions of the set-covering problem may consist of scattered reserves with little spatial coherence. This is particularly critical when these sites are surrounded by a relatively impermeable matrix of land uses and land cover types. Spatial considerations may be implemented into reserve selection models (see Williams et al. (2005) for a review), but commonly fall into the application range of reserve design models. An example is the applied downscaling tool SWOMP (Schleupner, 2009) used to delineate possible expansion areas of the Natura 2000 system (Chapter IV).

Closely linked with the spatial aspects of reserve design are uncertainties that result from using data at different resolutions. The coarse species occurrence data make it difficult to know whether species recorded in a particular planning unit are actually present inside corresponding protected areas (see also a study by Araujo (2004) on this problem). For this reason, a validation of the model results with independent datasets as outlined in Chapter IV is necessary.

Despite the given data and model deficiencies and inaccuracies, the application of these tools still seems essential. A major argument is that conservation planning should not be delayed until improved biodiversity data and planning tools are available. As biodiversity losses are principally irreversible, delayed conservation actions may leave fewer options for the future.

Nevertheless, another critical aspect needs to be mentioned. Given the importance placed on protected areas for the safeguarding of biodiversity, examining their performance in representing and maintaining biodiversity features is a central issue in systematic conservation planning and poses a major challenge for conservation biology (Gaston et al., 2006; Margules and Pressey, 2000). However, in isolation from other approaches, protected areas are plainly not sufficient for the conservation of biodiversity. The threats to biodiversity have to be addressed also beyond the borders of protected areas. Pressures from habitat loss, land-use change and degradation, and unsustainable water use have to be reduced. Threats from invasive alien species have to be controlled. Pollution through nutrient loading of nitrogen and phosphorus and other substances such as persistent organic pollutants has to be reduced. Furthermore, maintaining and enhancing the resilience of the components of biodiversity to adapt to climate change is crucial. Addressing all these challenges requires a sustainable development of mankind.

2 Outlook and concluding remarks

The studies introduced in this thesis may be complemented by further research in the fields of systematic conservation planning and the linkage of natural and social sciences.

A potential extension that could be implemented in the existing framework without major modifications is a more detailed representation of wetland habitats, including changes in extent and distribution due to climate change impacts. A cooperation with the research group 'Terrestrial Hydrology' of the Max Planck Institute for Meteorology on this topic is planned.

In Europe and globally, wetland ecosystems suffer from high nitrogen deposition rates leading to massive vegetational changes from eutrophication (Hogg et al., 1995; Pauli et al., 2002; Venterink et al., 2002). A correlation of priority areas for conservation – existing reserves as well as promising expansion sites - and regions with high pressures from nitrogen deposition would help to identify locations demanding immediate action for reducing nitrogen impact on wetland biodiversity.

One shortcoming of current economic land-use models from a biological perspective is that they do not explicitly consider biodiversity conservation as a further land-use option. An example for such a model is the European Forest and Agricultural Sector Optimization Model (EUFASOM, Schneider et al., 2008); a partial equilibrium, bottom-up land use model. An iterative linkage of the HABITAT model and EUFASOM would enable an integrated assessment of biodiversity and related environmental objectives. Synergies and tradeoffs between biodiversity, climate, energy, and other policies which affect land use could be explored.

Potential extensions of the introduced framework also include a widening of the model scope to other species and habitats of European conservation concern, e.g. the complete species and habitat assemblage of the Natura 2000 directives. Especially gap analyses would be more inclusive then. Another option is to work on smaller scales, e.g. address a single European country for which high resolution data on the distribution of biodiversity are available.

Reducing or even halting the loss of biodiversity is an extremely difficult task. This process will extend well beyond the year 2010. The past years of research have constantly contributed to strengthen important tools to plan for the safeguarding of biodiversity features. This thesis is one further contribution.

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Appendix 1

Table A1: Vernacular names of wetland species

	English	**German**
Amphibians		
Alytes muletensis	Mallorcan midwife toad	Balearen-Geburtshelferkröte
Bombina bombina	Fire-bellied toad	Rotbauchunke
Bombina variegata	Yellow-bellied toad	Gelbbauchunke
Chioglossa lusitanica	Golden-striped salamander	Goldstreifensalamander
Discoglossus galganoi	Iberian painted frog	Iberischer Scheibenzüngler
Discoglossus montalentii	Corsican painted frog	Korsischer Scheibenzüngler
Discoglossus sardus	Tyrrhenian painted frog	Sardischer Scheibenzüngler
Pelobates fuscus insubricus	Common spadefoot	Italienische Knoblauchkröte
Rana latastei	Italian agile frog	Italienischer Springfrosch
Salamandrina terdigitata	Spectacled salamander	Brillensalamander
Triturus carnifex	Italian crested newt	Alpen-Kammolch
Triturus cristatus	Great crested newt	Kammolch
Triturus dobrogicus	Danube crested newt	Donau-Kammolch
Triturus karelini	Southern crested newt	Balkankammmolch
Triturus montandoni	Carpathian newt	Karpatenmolch
Triturus vulgaris ampelensis[a]	Smooth newt	Rumänischer Teichmolch
Reptiles		
Elaphe quatuorlineata	Four-lined snake	Vierstreifennatter
Emys orbicularis	European pond tortoise	Europäische Sumpfschildkröte
Mauremys caspica	Stripe necked terrapin	Kaspische Wasserschildkröte
Mauremys leprosa	Spanish terrapin	Spanische Wasserschildkröte
Birds		
Acrocephalus paludicola	Aquatic warbler	Seggenrohrsänger
Alcedo atthis	Kingfisher	Eisvogel
Anser erythropus	Lesser white-fronted goose	Zwerggans
Aquila chrysaetos	Golden eagle	Steinadler
Aquila clanga	Spotted eagle	Schelladler
Ardea purpurea purpurea	Purple heron	Purpurreiher
Ardeola ralloides	Squacco heron	Rallenreiher
Asio flammeus	Short-eared owl	Sumpfohreule

Aythya nyroca	Ferruginous duck	Moorente
Botaurus stellaris stellaris	Bittern	Rohrdommel
Bucephala islandica[b]	Barrow's Goldeneye	Spatelente
Chlidonias hybridus	Whiskered tern	Weißbartseeschwalbe
Chlidonias niger	Black tern	Trauerseeschwalbe
Ciconia ciconia	White stork	Weißstorch
Ciconia nigra	Black stork	Schwarzstorch
Crex crex	Corncrake	Wachtelkönig
Fulica cristata	Crested coot	Kammbläßhuhn
Gavia arctica	Black-throated diver	Prachttaucher
Gelochelidon nilotica	Gull-billed tern	Lachseeschwalbe
Glareola pratincola	Collared pratincole	Brachschwalbe
Grus grus	Crane	Kranich
Haliaeetus albicilla	White-tailed eagle	Seeadler
Histrionicus histrionicus[b]	Harlequin duck	Kragenente
Hoplopterus spinosus	Spur-winged plover	Spornkiebitz
Ixobrychus minutus minutus	Little bittern	Zwergdommel
Marmaronetta angustirostris	Marbled teal	Marmelente
Milvus migrans	Black kite	Schwarzmilan
Nycticorax nycticorax	Night heron	Nachtreiher
Oxyura leucocephala	White-headed duck	Weißkopf-Ruderente
Pandion haliaetus	Osprey	Fischadler
Pelecanus crispus	Dalmatian pelican	Krauskopfpelikan
Pelecanus onocrotalus	White pelican	Rosapelikan
Phalacrocorax pygmaeus	Pygmy cormorant	Zwergscharbe
Philomachus pugnax	Ruff	Kampfläufer
Platalea leucorodia	Spoonbill	Löffler
Plegadis falcinellus	Glossy ibis	Braunsichler
Porphyrio porphyrio	Purple gallinule	Purpurhuhn
Porzana parva parva	Little crake	Kleines Sumpfhuhn
Porzana porzana	Spotted crake	Tüpfelsumpfhuhn
Porzana pusilla	Baillon´s crake	Zwergsumpfhuhn
Sterna albifrons	Little tern	Zwergseeschwalbe
Tadorna ferruginea	Ruddy shelduck	Rostgans
Tringa glareola	Wood sandpiper	Bruchwasserläufer

Mammals

Castor fiber	Eurasian beaver	Europäischer Biber
Galemys pyrenaicus	Pyrenean desman	Pyrenäen-Desman
Lutra lutra	European otter	Fischotter
Microtus cabrerae	Cabrera's vole	Cabreramaus
Microtus oeconomus arenicola	Dutch root vole	Niederländische Wühlmaus
Microtus oeconomus mehelyi	Pannonian root vole	Ungarische Wühlmaus
Mustela lutreola	European mink	Europäischer Nerz
Myotis capaccinii	Long-fingered bat	Langfußfledermaus
Myotis dasycneme	Pond bat	Teichfledermaus

[a] The genus *Triturus v. ampelensis* does only occur in Romania and is therefore not included in the analyses of Chapter II with a spatial scope of EU25.
[b] *Bucephala islandica* and *Histrionicus histrionicus* only occur on Iceland and are therefore solely included in the analyses of Chapter III with a European-wide spatial scope.

Appendix 2

Table A2: Literature sources of ecological model input data

Taxon	Population density	Habitat type requirements
Amphibians	Due to data deficiencies regarding most of the amphibian species' densities, the values were estimated to 10 reproductive units per hectare for solitary species and 20 reproductive units per hectare for gregarious species.	AmphibiaWeb (2007), IUCN (2007)
Reptiles	Böhme (1981-)	Böhme (1981-)
Birds		Hagemeijer and Blair (1997), Tucker and Evans (1997)
Acrocephalus paludicola	Hagemeijer and Blair (1997)	
Alcedo atthis	Hagemeijer and Blair (1997)	
Anser erythropus	Cramp (1992), Hearn (2004)	
Aquila chrysaetos	Hagemeijer and Blair (1997)	
Aquila clanga	Hagemeijer and Blair (1997)	
Ardea purpurea purpurea	Cramp (1992) *	
Ardeola ralloides	Cramp (1992) *	
Asio flammeus	Hagemeijer and Blair (1997)	
Aythya nyroca	Snow and Perrins (1998-), Niethammer (1966-)	
Botaurus stellaris stellaris	Hagemeijer and Blair (1997)	
Bucephala islandica	Einarsson et al. (2006), Hagemeijer and Blair (1997)	
Chlidonias hybridus	Cramp (1992) *	
Chlidonias niger	Snow and Perrins (1998-), Cramp (1992)	
Ciconia ciconia	Denac (2006)	
Ciconia nigra	Hagemeijer and Blair (1997)	
Crex crex	Snow and Perrins (1998-), Cramp (1992)	
Fulica cristata	Snow and Perrins (1998-), Cramp (1992)	
Gavia arctica	Hagemeijer and Blair (1997)	
Gelochelidon nilotica	Cramp (1992) *	
Glareola pratincola	Cramp (1992)	
Grus grus	Hagemeijer and Blair (1997)	

Scientific name	Population density	Habitat type requirements
Haliaeetus albicilla	Hagemeijer and Blair (1997)	
Histrionicus histrionicus	Einarsson et al. (2006), Hagemeijer and Blair (1997)	
Hoplopterus spinosus	Cramp (1992), Hagemeijer and Blair (1997)	
Ixobrychus minutus minutus	Cramp (1992), Hagemeijer and Blair (1997)	
Marmaronetta angustirostris	Snow and Perrins (1998-), Cramp (1992) *	
Milvus migrans	Snow and Perrins (1998-), Hagemeijer and Blair (1997)	
Nycticorax nycticorax	Hagemeijer and Blair (1997), Kazantzidis et al. (1997)	
Oxyura leucocephala	Hagemeijer and Blair (1997)	
Pandion haliaetus	Hagemeijer and Blair (1997)	
Pelecanus crispus	Snow and Perrins (1998-) *, Catsadorakis and Crivelli (2001)	
Pelecanus onocrotalus	Snow and Perrins (1998-) *, Catsadorakis and Crivelli (2001)	
Phalacrocorax pygmaeus	Volponi (1999), Vaneerden and Gregersen (1995) *	
Philomachus pugnax	Hagemeijer and Blair (1997)	
Platalea leucorodia	Cramp (1992), BirdLife International (2001) *	
Plegadis falcinellus	Snow and Perrins (1998-), Parsons (1995) *	
Porphyrio porphyrio	Hagemeijer and Blair (1997)	
Porzana parva parva	Hagemeijer and Blair (1997)	
Porzana porzana	Hagemeijer and Blair (1997)	
Porzana pusilla	Cramp (1992)	
Sterna albifrons	Snow and Perrins (1998-), Cramp (1992) *	
Tadorna ferruginea	Snow and Perrins (1998-), Young (1970)	
Tringa glareola	Hagemeijer and Blair (1997)	
Mammals		Mitchell-Jones et al. (1999), Niethammer and Krapp (1978-)
Castor fiber	Niethammer and Krapp (1978-), Mitchell-Jones et al. (1999)	
Galemys pyrenaicus	Niethammer and Krapp (1978-), Mitchell-Jones et al. (1999)	
Lutra lutra	Niethammer and Krapp (1978-)	
Microtus cabrerae	Fernandez-Salvador et al. (2005)	

Scientific name	Population density	Habitat type requirements
Microtus oeconomus arenicola	Mitchell-Jones et al. (1999)	
Microtus oeconomus mehelyi	Mitchell-Jones et al. (1999)	
Mustela lutreola	Niethammer and Krapp (1978-)	
Myotis capaccinii	Niethammer and Krapp (1978-), Robinson and Stebbings (1997)	
Myotis dasycneme	Niethammer and Krapp (1978-), Robinson and Stebbings (1997)	

* Due to data deficiencies regarding the densities on foraging areas for colonial birds, nesting and foraging areas are differentiated. The foraging area is set to 5 hectares per reproductive unit.

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