Max-Planck-Institut für Meteorologie
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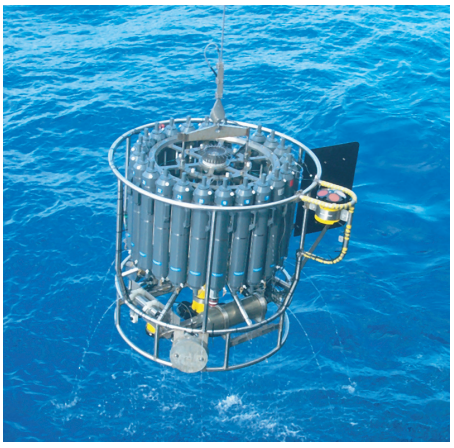
GIS as integrating tool in Sustainability and Global Change

Christine Schleupner



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Christine Schleupner

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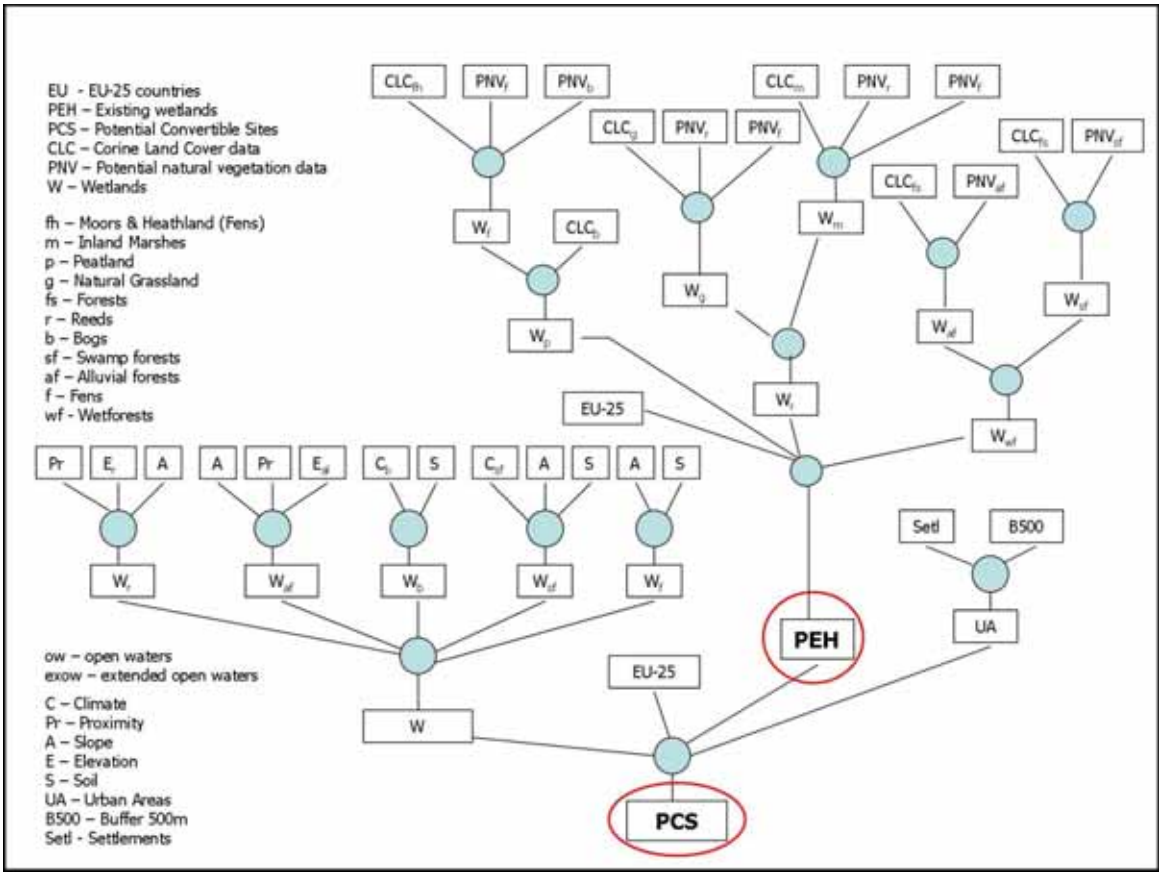

Christine Schleupner

Hamburg 2008

Ein dicker Sack - den Bauer Bolte,
Der ihn zur Mühle tragen wollte,
Um auszuruhn, mal hingestellt
Dicht an ein reifes Ährenfeld -
Legt sich in würdevolle Falten
Und fängt 'ne Rede an zu halten.
Ich, sprach er, bin der volle Sack.
Ihr Ähren seid nur dünnes Pack.
Ich bin's, der euch auf dieser Welt
In Einigkeit zusammenhält.
Ich bin's, der hoch vonnöten ist,
Daß euch das Federvieh nicht frißt;
Ich, dessen hohe Fassungskraft
Euch schließlich in die Mühle schafft.
Verneigt euch tief, denn ich bin Der!
Was wäret ihr, wenn ich nicht wär?
Sanft rauschen die Ähren:
Du wärst ein leerer Schlauch,
wenn wir nicht wären.

Wilhelm Busch (1832 - 1908)

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The GIS-Software ArcGIS 9 has been kindly sponsored by the ESRI-Absolventenprogramm. I would like to thank various research institutions for sharing valuable data and essential material. This study would not have been possible without the provision of their data. Special thanks to Dr. E. Tschach of the Landesamt für Natur und Umwelt des Landes Schleswig-Holstein.

The contents of this thesis have been greatly improved by the comments of a number of anonymous referees. The thesis also benefits from the attendance of interesting conferences and summer schools and the discussions, acquaintance, and friendships that evolved with experts of various disciplines.

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Preface

This thesis is the result of approximately five years' research on different subjects and at different scales. Therefore, this thesis does not originate from a single, well-defined PhD project but rather combines and integrates results of a number of different studies that at first seem only loosely related. According to our Research Group's aims they can be best summarized under the topic "Applications of GIS in Sustainability and Global Change".

The cumulative studies presented here contribute to current discussions of the impacts of climate change on land use. The investigated impacts of climate change are in this case direct through sea level rise or hurricanes etc., or are indirect through promotion of competing land use options as bioenergy plantations for greenhouse gas mitigation options, for example. In the foreground stand assessments of the consequences to social, economic or ecological sustainability of the landscape function caused by these climate change induced impacts. Land use change and climate change have important effects on species and ecosystems, but also on the sustainable supply of resources and crucial ecosystem services to humanity (EHRLICH 2007). In general, the term sustainability encompasses economic, environmental, and social issues simultaneously what has often been ignored in the past. It is therefore increasingly recommended to connect social and physical geosciences more strongly with focus on human-environmental interactions (LÖFFLER & STEINHARDT 2004; LINDENMAYER & HOBBS 2007). The common theme of this thesis is the analysis of interactions between land use change and the environment with a focus on geographical border areas such as coastal regions or those intermediate areas called wetlands, which lie between terrestrial land and aquatic systems. The focus is not only on the natural environment, but also on human environments. Within this respect, humans need to be considered as part of the application, but neither as the underlying problem nor in anthropocentric terms only.

The definition and application of goals is an important subject of this thesis. How a landscape is defined, characterized and then classified can have a significant effect on a wide range of management decisions. Different problems, objectives and goals require different classifications. Therefore, it is necessary to clearly state the objectives. One chapter deals with the formulation of adaptation strategies to sea level rise; in another targets for the location of optimal wetland restoration sites are defined, or the introduction of bird conservation areas is

questioned. Within this respect, there is a need for the conduction of prioritization frameworks that allow decisions on what to do and where to do it. On that score, spatial issues are extremely important to land use planning. Also emerging landscape-scale processes that affect large numbers of species make spatially explicit strategies essential (BURGMAN ET AL. 2007). E.g., the identification of spatially explicit options for improving connectivity etc. is often requested (SCOTT & TEAR 2007). Geographic Information Systems (*GIS*) proved to be an essential tool to analyse the different issues applied in this thesis. GIS is an analysis tool and has the ability to manipulate multiple data layers, providing a systematic and dynamic approach. There is growing interest in GIS solutions and in linking other models to GIS. But still the potential of GIS is by far greater than its actual utilization (c.f. LANG & BLASCHKE 2007 and LONGLEY ET AL. 2002 for a history of GIS). Nevertheless, GIS is seen as most important tool for landscape analysis, planning and management (BASTIAN & STEINHARDT 2002). The new-acquisition of information serves as support for spatial decisions. GIS offers valuable contributions to support the more complex growing planning tasks through the explanation of spatial context. It allows the development of spatial scenarios, and the evaluation of optimal variants through modelling of different factor combinations (LANG & BLASCHKE 2007). This helps to quickly identify different options for decision making. Maps simplify the communication of visual and quantitative information to organisations.

The thesis is divided into four parts, of which the first three parts contain the studies undertaken whereas the fourth part reflects on the studies and provides an outlook for future tasks. Part one deals with the evaluation of impact potentials to the coast of Martinique caused by extreme weather events and sea level rise. Part two looks at the consequences of land use changes for bird populations. This time the Eiderstedt peninsula in Schleswig-Holstein/Germany is in the centre of interest. Subsequently, the third part considers Europe at a larger scale and discusses the development of a spatial wetland distribution and optimal site selection model for different policy options in EUFASOM (European Forest and Agricultural Sector Optimization Model). Table 0.1 gives an overview of the publication status of the following chapters.

Table Author, journal and status of papers of the thesis.

Ch.	Title	WP No	Author	Journal
1	Evaluating the regional impact potential to erosion and inundation caused by coastal hazards	FNU 152	SCHLEUPNER	In review
2	Evaluation of coastal squeeze and its consequences for the Caribbean island Martinique		SCHLEUPNER	Ocean and Coastal Management 51 (5), pp. 383-390
3	Spatial assessment of sea level rise on Martinique's coastal zone and analysis of planning frameworks for adaptation		SCHLEUPNER	Journal of Coastal Conservation: Planning & Management 11 (2), pp. 91-103.
4	Agricultural land use changes in Eiderstedt: historic developments and future plans		LINK & SCHLEUPNER	Coastline Reports 9, pp. 197-206
5	Potential impacts on important bird habitats in Eiderstedt (Schleswig-Holstein) caused by agricultural land use changes		SCHLEUPNER & LINK	Applied Geography 28 (4), pp. 237-247
6	Estimation of spatial wetland distribution potentials in Europe	FNU 135	SCHLEUPNER	In review
7	Evaluation of European wetland restoration potentials by considering economic costs under different policy options	FNU 158	SCHLEUPNER & SCHNEIDER	In review
8	A cost-efficient spatial site-selection model for European wetland restoration	FNU 159	SCHLEUPNER & SCHNEIDER	In review

SUMMARY

Rapid land use changes and the impacts of climate change are seen as a major threat to biodiversity preservation and the supply of crucial ecosystem services to humanity. This thesis contributes to actual discussions of the direct and indirect impacts of climate change and climate mitigation politics to land use. It is divided into three parts:

The main topic of **Part one** is the evaluation of impact potentials to the coast of the Caribbean island Martinique caused by extreme weather events and sea level rise. The **first chapter** deals with the development of a GIS-based model for the island that evaluates the sensitivity of the coastal areas to erosion, flooding and inundation. This includes an analysis of the extension of the potential impact area. The results are illustrated in sensitivity and hazard area maps for the Martinique coast that serve as base for further vulnerability studies. In the **second chapter** the Martinique beaches and coastal wetlands are examined to identify the probability of coastal squeeze. In many cases coastal development prevents coasts from adapting to accelerated sea level rise by shifting landward. Also tourism infrastructure augments the probability of beach reduction and mangrove squeeze. On the mountainous island Martinique the majority of settlements and especially tourist hotels are built within the zone at risk to flooding and erosion. Spatial analysis based on a conducted GIS model is carried out that evaluates the tourist destinations most vulnerable to the impacts of sea level rise. If sea level rises and beach reduction becomes an increasing problem the attractiveness of Martinique beaches as tourist destination is likely to decline. **Chapter 3** deals with the evaluation of human vulnerability to accelerated sea level rise on the Martinique coast. In addition, the possible effects of sea level rise on the island are spatially assessed for future regional planning purposes. The actual situation and legislation measures for coastal zone management of the island are described and sea level rise response strategies are discussed. Even if saltwater intrusion and coastal erosion with increasing offshore loss of sediment are locally already a severe problem, potential rises in sea level and its impacts are not addressed in coastal management. This chapter sees itself as recommendation of action not only for Martinique.

Part two deals with the impacts of land use changes for bird populations on the Eiderstedt peninsula in Schleswig-Holstein (Germany). In the past, the landscape has been generally dominated by extensively used grassland. These grassland areas are home to many bird species, and among naturalists Eiderstedt is considered to be one of the prime bird habitats in Schleswig-Holstein. Ongoing changes in the structure of the regional agriculture towards an intensified cattle breeding and the growth of biofuels call for a conversion of large shares of grassland to arable farm land. At the same time a fiercely debate arose to what extent Eiderstedt can be declared as bird conservation area within the Natura 2000 network. In **chapter 4** the drivers and accompanying conflicts of rapid land use changes on Eiderstedt are explored in more detail. Under consideration of

the regional land utilization history three possible scenarios of transformations of agricultural land are developed which can be applied to determine the possible impacts of such conversions. In **chapter 5**, the possible impacts of agricultural land use change on Eiderstedt on breeding bird populations of four key species are determined. The results indicate that an increase of arable farm land to approximately two thirds of the whole agricultural area drastically reduces suitable bird habitat, thus considerably diminishing the number of breeding pairs supported by the environment.

The **third part** evaluates potentials to preserve existing habitats, to restore formerly native habitats, as well as to create non-native managed habitats with respect to freshwater wetlands of the EU. **Chapter 6** deals with the methodological development of a spatial wetland distribution model (SWEDI) and description of its results. Through the GIS-based model the spatial extent of existing wetland distribution within the EU-25 countries is visualised. Additionally, potential convertible sites are modelled for (re-) creation of wetland biotopes. Because the existence of wetlands is driven by site specific natural conditions and the economic environment, **chapter 7** integrates both aspects by linking the GIS-based wetland model with the European Forest and Agricultural Sector Optimization Model (EUFASOM). EUFASOM is a partial equilibrium model which studies simultaneously synergies and tradeoffs between biodiversity conservation efforts, greenhouse gas mitigation options including carbon sinks and bioenergy, as well as traditional agriculture and forestry markets. For different policy scenarios, the optimization model computes the corresponding economic potential of wetlands, its effects on agricultural and forestry markets, and environmental impacts. In **chapter 8** the scenario specific total wetland area per EU-country from EUFASOM is downscaled by a GIS-based site-selection model which uses environmental constraints. The final result is a wetland site-selection model that evaluates optimal distributions for wetland restoration sites levelled after defined restoration goals and dependent on the EUFASOM scenarios. The model is useful to locate sites suitable for renaturational programs and for the effective introduction of faunistic corridors considering the NATURA 2000 network.

In all, the studies show that the inclusion of GIS-based assessment tools is essential to favour effective regional conservation planning and to improve the predictive capacities of coastal zone management plans. Often the illustration of scientific results through maps is indispensable to support public participation. Applying GIS solutions in sustainability and global change helps filling the still existing gap between social sciences and physical geosciences.

ZUSAMMENFASSUNG

Diese Arbeit leistet einen Beitrag zu aktuellen Diskussionen über die direkten und indirekten Folgen des Klimawandels und der Klimapolitik auf unterschiedliche menschliche Landnutzungsansprüche. Um Lösungsansätze zu formulieren, werden Konfliktfelder zwischen Natur und Mensch räumlich analysiert und beurteilt. Vorliegende Arbeit gliedert sich in drei Teile:

Im **ersten Teil** dieser Arbeit werden mögliche Folgen von extremen Wetterereignissen und eines beschleunigten Meeresspiegelanstieges auf die Küste Martiniques ermittelt. Das **erste Kapitel** beschäftigt sich mit der Entwicklung eines GIS-basierten Modells der Insel Martinique, welches die Erosions- und Überschwemmungs-Sensitivität der Küstengebiete bestimmt und auch eine Analyse des potentiellen Risikogebietes beinhaltet. Die daraus resultierenden Sensitivitäts- und Risikogebietskarten dienen als Basis für anschließende Vulnerabilitätsstudien. Im **zweiten Kapitel** werden Martiniques Strände und Mangroven hinsichtlich ihrer Gefährdung durch „Coastal Squeeze“ analysiert. Auf der gebirgigen Insel Martinique liegt die Mehrzahl der Siedlungen entlang der flachen Küstenabschnitte nahezu auf Meeresspiegelhöhe. In vielen Fällen verhindert die Besiedlung und insbesondere die touristische Infrastruktur eine natürliche Anpassung der Küste an einen Meeresspiegelanstieg. Der Verlust von Mangroven stellt ein Sicherheitsproblem dar, da diese dem Hinterland als Überflutungsrückhalt dienen. Erhebliche Strandverluste durch Erosion könnten ebenfalls die Attraktivität der Insel als Touristenziel mindern. **Kapitel 3** beschäftigt sich mit Vulnerabilitätsstudien hinsichtlich gesteigerter Küstenerosion und Überflutung verursacht durch einen steigenden Meeresspiegel. Zudem werden mögliche Folgen eines Meeresspiegelanstieges für zukünftige Regionalplanungsvorhaben bewertet und Response Strategien diskutiert, denn obwohl Salzwasserintrusion und Küstenerosion schon lange ein bedeutendes Problem auf Martinique darstellen, werden die Folgen eines Meeresspiegelanstiegs im lokalen Küstenzonenmanagement nicht beachtet. Dieses Kapitel sieht sich als Handlungsempfehlung nicht nur für Martinique.

Der **zweite Teil** dieser Arbeit beinhaltet Untersuchungen zu den Folgen von Landnutzungsveränderungen auf Eiderstedter Vogelpopulationen (Schleswig-Holstein/ Deutschland). In der Vergangenheit wurde das Landschaftsbild auf der Halbinsel in der Regel von extensiv genutztem, feuchtem Weideland dominiert. Diese Grasländer sind Heimat und Rastgebiet vieler Vögel und Eiderstedt gilt unter Naturfreunden als einer der wichtigsten Lebensräume für Wiesenvögel in Schleswig-Holstein. Intensive Milchwirtschaft, zunehmende Stallmast mit einhergehendem verstärktem Kraftfutteranbau sowie der Anbau von Bioenergiepflanzen führt in jüngster Zeit jedoch zu einem anhaltenden Grünlandumbruch in der regionalen Landwirtschaft hin zu intensiver Ackerwirtschaft. Gleichzeitig läuft eine heftige Debatte, inwieweit Eiderstedt als Vogelschutzgebiet im Rahmen von Natura 2000 deklariert werden kann. Im **vierten**

Kapitel wird die Landnutzungsgeschichte Eiderstedts sowie die Hintergründe dieses Konfliktes beleuchtet und die möglichen ökologischen Konsequenzen eines Grünlandumbruchs mittels Szenarien aufgezeigt. Darauf aufbauend beschäftigt sich **Kapitel 5** mit den Folgen des Grünlandumbruchs auf Eiderstedt für die Populationen von vier ausgewählten repräsentativen Wiesenbrutvogelarten. Ergebnisse indizieren, dass eine Zunahme von Ackerland auf etwa zwei Drittel der gesamtlandwirtschaftlichen Fläche nicht nur drastisch das geeignete Bruthabitat reduziert, sondern gleichzeitig auch die Qualität verbleibender Standorte schmälert.

Im **dritten Teil** dieser Arbeit werden sowohl existierende Süßwasser beeinflusste Feuchtgebiete der EU als auch potentielle Restaurationsflächen mittels eines GIS-Modells räumlich explizit ermittelt und hinsichtlich optimaler ökonomischer und ökologischer Standorte analysiert. **Kapitel 6** handelt von der methodischen Entwicklung eines Feuchtgebietsverteilungsmodells (SWEDI) im Europäischen Maßstab und dessen Illustration. Durch das GIS-basierte Modell können sowohl die räumliche Verbreitung existierender Feuchtgebietsflächen dargestellt, als auch potentielle Feuchtgebietsrenaturierungsflächen ermittelt werden. Da die Existenz von Feuchtgebieten von ortsspezifischen physisch-geographischen Faktoren aber auch von der ökonomischen Umwelt bestimmt wird, werden in **Kapitel 7** beide Aspekte miteinander verbunden, indem ein GIS-basiertes Modell mit dem Europäischen Forst- und Landwirtschaftssektor-Optimierungsmodel (EUFASOM) verknüpft wird. EUFASOM analysiert gleichzeitig Synergien und Tradeoffs zwischen Land- und Forstwirtschaft, Biodiversitätsschutz sowie Treibhausgasvermeidungsstrategien inklusive Kohlenstoffsenken und Bioenergie. Für verschiedene politische Szenarien berechnet das Modell das ökonomische Potential von Feuchtgebieten sowie dessen Einfluss auf den Land- und Forstwirtschaftsmarkt. In **Kapitel 8** wird die szenarienabhängige, spezifische Gesamtfeuchtgebietsfläche pro EU-Staat aus EUFASOM mit Hilfe eines GIS-basierten Gebietsauswahlmodells räumlich explizit ausgewertet. Das Endergebnis ist ein Feuchtgebietsauswahlmodell, das optimale Verteilungen von Feuchtgebiets-renaturierungsflächen ermittelt, indem es Prioritäten aus Renaturierungszielen setzt. Das Modell ist nützlich bei der überregionalen Ermittlung geeigneter Flächen für Renaturierungsprogramme oder für die Einführung von faunistischen Korridoren im Sinne von Natura 2000. Gleichzeitig kann es zum Erfolg regionaler Naturschutzplanung beitragen.

GIS Anwendungen haben Konfliktlösungspotential bei Problemfeldern in Nachhaltigkeit und Klimawandel, aber sie sind ebenso essentiell bei Bemühungen die Kluft zwischen Sozial- und physischen Geowissenschaften zu schließen und die Forschungsergebnisse auf verständliche Weise einer breiteren Öffentlichkeit zugänglich zu machen.

Part I

Spatial Analysis of Coastal Impacts on Martinique



Introduction

This study chose the mountainous Caribbean island Martinique as case study site. Martinique is a French island of the Lesser Antilles in the Caribbean region. Figure a gives an overview of the Caribbean region with Martinique.

Fig a Overview of the Caribbean islands and position of Martinique.

The tropical climate of the region is moderated by trade winds, and the rainy season lasts from July to December. During that season floods often occur which are mainly caused by hurricanes. The geology of Martinique is dominated by volcanoes of different age. The island evolved over the last 20 million years because of eruptions of volcanoes that were displaced northwards due to tectonic movements. The youngest volcano is still active. It is the Mt. Pelée (1 396 m) situated in the north of the island. Superficially, the island has mountainous character with numerous but small rivers.

Martinique is a French Department and EU „ultra-peripheral region“. The economy is largely based on the export of agricultural goods (bananas, sugarcane, and pineapples) and tourism as major income sources. Nearly one million visitors annually arrive on the island which is inhabited by approximately 400 000 people (MARQUES 2002; CHARRIER 2003). The majority of tourists stay in hotels on the coast particularly in the southern part of Martinique. Because the topography of the island is characterised by steep mountains the majority of the settlements and of the population are situated along the coast below 20 m.

Figure b gives an overview of the first part which results in the first three chapters of this thesis. The first chapter deals with the evaluation of the impact potential of the coast to erosion and inundation, whereas the second chapter describes in detail the development of a methodology to evaluate the coastal squeeze phenomenon due to sea level rise and its application to Martinique’s coastal wetlands and beaches. In the third chapter then

erosion and inundation areas are evaluated based on certain sea level rise projections and the resulting human vulnerability including adaptation strategies.

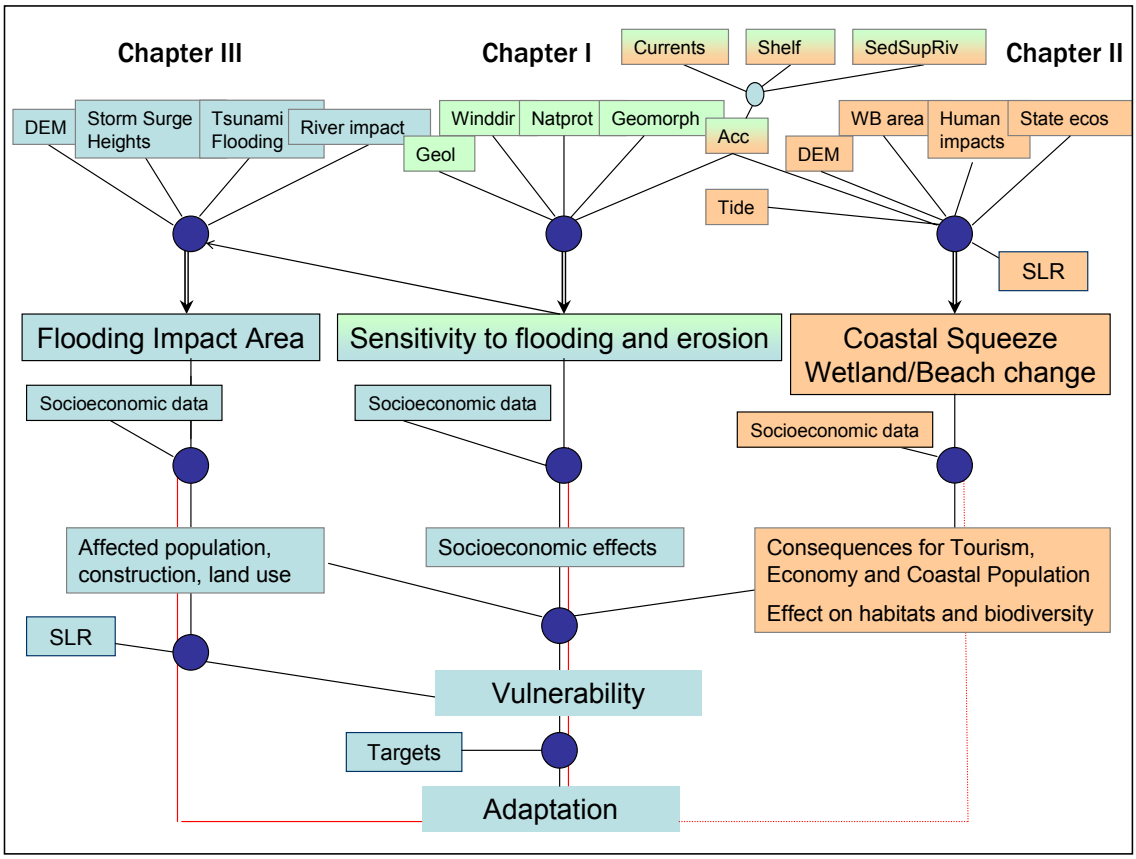


Fig b Overview of the methodological structure of the coastal assessment that is subdivided into the three chapters 1 (green), 2 (orange), and 3 (blue).

1. Evaluating the Regional Impact Potential to Erosion and Inundation Caused by Coastal Hazards

1.1. Introduction

The main purpose of this study is to evaluate the coastal sensitivity to flooding and erosion and its potential impact areas caused by extreme storm events. Coastal hazards do not only influence the morphology of a coast, but also strongly impact human coastal resources. Therefore it is not surprising that numerous studies about the impacts of hurricanes exist. The majority of scientific studies concentrate on post-disaster management evaluation. Recent studies deal with the hurricane events "Ivan" and "Katrina" and assess their impacts on vegetation recovery and forest resources (BURLEY ET AL. 2007; OSWALT & OSWALT 2008), on sedimentation and geomorphology (CLAUDINO-SALES ET AL. 2008; SILVERBERG ET AL. 2008), but also on infrastructure, energy and health (LEWSEY ET AL. 2004; BAYLEYEGN ET AL. 2006; LI & ELLINGWOOD 2006). STOCKDON ET AL. (2007) introduce a model for coastal response to hurricanes; CHEUNG ET AL. (2003) and COLBY ET AL. (2000) simulate storm-induced coastal flooding, for example. The aim of this study is to present a more farsighted component of hazard evaluation by integrating spatial analyses into coastal sensitivity assessments. The study presented here focuses on the Caribbean Lesser Antilles island Martinique. On average, storms stronger than Category 3 are passing close to any given island in the Caribbean every two to five years. An evaluation by METEO-FRANCE (2000) state that a cyclonic phenomenon (hurricane or tropical storm) occurs every 3.6 years on Martinique, one serious hurricane on average every 13 years. From 1886 to 2005, Martinique was hit by 53 storms 19 of them categorized as hurricanes with wind speeds of more than 118 km/h. These storms have nearly always caused extensive damage to the human coastal resources. In 1780, about 22 000 people in the Lesser Antilles were killed by a hurricane – mainly on Martinique and Barbados (ZAHIBO ET AL. 2007). Wave heights of more than six meters are common with erosion and inundation as its consequences (GONZALES 1988). Many islands and coastal states within the hurricane impact

area prepare for these hazards by conducting emergency and evacuation plans. The latest IPCC report (PARRY ET AL. 2007) and regional climate change projections for the Caribbean region (UNEP 2000) expect an increasing frequency and intensity of hurricanes and tropical storms that coincide with coastal flooding and high erosion rates at the shores (MAUL 1993). With increasing hurricane activity the impact area extent will also increase. However, not only these extreme storm events but also waves caused by earthquakes need to be considered in coastal hazard impact evaluation. Earthquakes within the Caribbean region are not rare due to the proximity of the subduction zone of the Atlantic under the Caribbean plate. Tsunamis are therefore an omnipresent risk in the region.

The coastal zone of Martinique is a very diverse space, partly occupied by settlements, used as famous tourist destination, or covered with valuable ecosystems. Most of the island's settlements are situated along the coast and beach tourism is the main income source. Even if local emergency systems exist and the responsible public authorities have published hazard maps (PERRET ET AL. 1996), there is still a need for vulnerability studies. Often these assessments require detailed and complete data. This has limited their applicability to other coasts and made the development of alternative assessment methodologies necessary. Consequently, this has led to a lack of vulnerability studies in areas where accurate quantitative data are missing (KLEIN & NICHOLLS 1999). There is a need for vulnerability studies along those coasts where the data availability is poor (KLEIN & NICHOLLS 1999; DEEB 2002). The objective of this case study is to provide a GIS-based methodology that only relies on easily obtainable spatial data and at the same time is transferable to other coasts. The available hazard data are an important requisite for model validation.

1.2. Materials and Methods

1.2.1. Sensitivity assessment of the Martinique coast to erosion and inundation

The determination of the sensitivity of a coast to the impacts of hazards is an important step in human vulnerability assessments. In these analyses the spatial component plays a vital role. The coastal sensitivity to flooding and erosion and the potential extent of the impact area caused by coastal hazards is evaluated

using a conceptual GIS-based model. The empiric rule-based approach presented here aims to localize those coastal strips that are especially affected by erosion and inundation during hurricanes using a GIS-environment. Figure 1.1 shows the logical decision tree on which the model is based on.

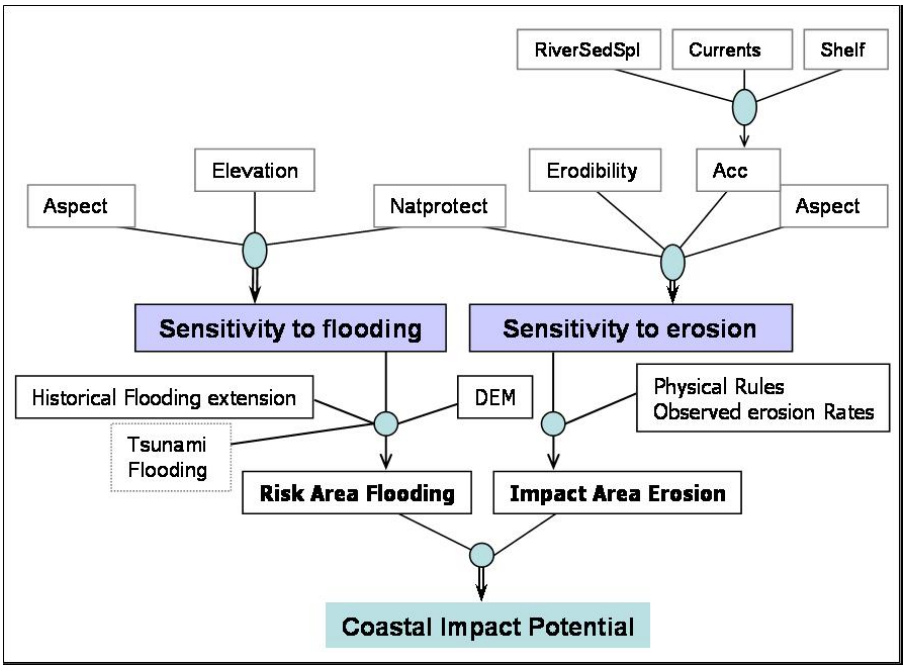


Fig 1.1 Overview of the methodological structure of the GIS-based model.

First of all, a relational geo-database is created including the most important factors that illustrate the nature of the coast as well as its coastal function. To evaluate the relation between coastal parameters and its sensitivity to erosion and inundation, data on Caribbean hurricane events and its impacts are correlated to the geo-database. Data on cyclones that passed over Martinique and their impacts can be found in METEO-FRANCE (2000), SAFFACHE ET AL. (2002), CARIBBEAN HURRICANE NETWORK. But also impact studies of other Caribbean islands with similar features have been used as surrogates for missing local data. These data are used for multiple regression analyses to determine those parameters that influence the coastal sensitivity to flooding and coastal erosion during extreme storm events most (cf. GORNITZ ET AL. 1997; FRIHY ET AL. 2004; STONE & OXFORD 2004; DOMINGUEZ ET AL. 2005; ZAHIBO ET AL. 2007). This way not only the impacts of storm direction and intensity to the coastal data have been evaluated but also the correlation of erosion rates per bedrock, coastal morphology, exposure, land use, or elevation, for example. The erosion rate is on

the one hand determined by storm intensity, but moreover by the relative rock resistance to erosion, by the exposure of the coast to the storm through aspect and (natural) protection, and also by the potential to recover through accumulation processes. The area of flooding depends on storm intensity and exposure as well but also on relative coastal elevation. These parameters build the basis input of the model. Below, these five parameters and their basis data are described in more detail:

Relative elevation. The coastward elevation of the land is important for inundation studies. STOCKDON ET AL. (2007) create a model for the coastal response to hurricanes. They conclude that the relative elevations of storm-induced water levels can be compared to beach or dune heights to estimate the coastal responses. This implies the simple rule that the lower the coastal topography, the higher is its sensitivity to inundation. The elevation categorization is determined through coastal morphology and topography based on data by HINNEWINKEL & PETIT (1975), IGN (1996), LANDSAT.

Erodibility. Quantitative comparable data about coastal erodibilities for the Martinique rocks do not exist. To evaluate the sensitivity of the coast to erosion the erodibility attribute is therefore based on the geology variable of the database. It is derived from the geologic map of Martinique (GRUNEWALD 1961) and based on knowledge about the comprising rock resistance (HOOKE & ROHRER 1977; ANNADALE & KIRSTEN 1994). For that reason, only relative statements about the resistance of each rock type to physical and chemical weathering are made, based on the relative hardness of the minerals comprising the rock. In reality, a wider range of erodibilities exists for each single rock type depending on mineral content, cementation, grain size, and presence of planar elements (e.g. fractures) within the rock (ANNADALE 1995). Table 1.1 lists the erodibility ratings for the Martinique coastal geology that is described in short here. For more information, refer to GRUNEWALD (1965). The coastal rocks of Martinique consist mainly of young unconsolidated volcanic material that is highly erosive. At the north-eastern coast, the main building rock is pumice interrupted by breccias and heat-tuff with a small band of alluvium. The heat-tuffs of the 1902 eruption have already been washed out completely (WEYL 1966). Along the south-west coast less erosive lava flows and tuffs dominate the geology, while in the south and

along the east coast breccias and weathered volcanite of medium erodibility are the main rocks to be found along the coast with extensive fluvial sedimentation at river mouths, tuffs, and isolated tertiary volcanic cones in change. Small amounts of tertiary limestone occur only in the outer southeast island. Its erodibility is moderate. Alluvial sediments dominate the bay of Fort-de-France. They are categorized highly erosive. But generally, mangrove forests are found on these alluvial sediments that foster the accumulation (WOODROFFE 1992; AUGUSTINUS 1995). Under natural conditions mangrove stands recover fast after cyclone induced disturbances (PALING ET AL. 2007) because the loss of sediment is not severe. To prevent an overestimation of the erosion factor at mangrove stands, the subcategory "mangrove vegetation" is added to the category "erodibility" as supplementation. If mangrove forest stands can be found on alluvial ground, the erosion attribute is reduced to medium erodibility. This decision is based on the vegetation map of PORTECOP (1971), satellite data (LandSat), as well as topographical maps (IGN 1996).

Natural shelter of the coastal segments (Natprotect). The natural protection of the coastal segments, if sheltered by an island, reef or inside a bay, is taken into consideration in the analysis of this study, because the protected position might preserve the coast from high waves and erosive swells. Coral reefs have the function of natural breakwaters along the coastline (FRIHY ET AL. 2004; DOMINGUEZ ET AL. 2005). Where coral reefs serve as shelters the marine erosion is therefore less severe, but only under the condition of healthy reefs. The most extensive reef structures can be found along the eastern coast of Martinique, south of St. Marie. Many smaller islands are situated in front of the eastern main island coast as well. Smaller reefs form also a barrier towards the sea at the southern coast with the most extensive reef formations at St. Luce. In the shelter of the cays lie extensive seagrass-beds, and mangrove forests are found on the shore. Some isolated reef structures also occur in the bay of Fort-de-France. But most of the coral formations inside Fort-de-France bay are highly degraded and in a critical state due to pollution (SMITH ET AL. 1999; PAULIN 2002). In the vicinity of the west and north coast little intermittent reef structures or isolated reefs can be found, but they can not protect the coast in so far as they are too small and have been damaged because of past volcanic activity.

Coastal exposure to the wind regimes (Aspect). Cliff retreat rates are generally higher on windward coasts where wind and wave action is more intense (MORTON & SALLENGER 2003; FRIHY ET AL. 2004; FERREIRA 2005). The windward parts of the Martinique coast are therefore more sensitive to erosion and inundation. Data on storm tracks of the last 155 years are used from the CARIBBEAN HURRICANE NETWORK to evaluate the main storm direction. On Martinique the cyclones track generally runs from east to west or from southeast to northwest. The mean wave heights are higher along the eastern coast than on the leeward parts. During hurricanes the swell at the Atlantic coast, which is on average between 1.2 to 2.5 m, reaches up to five to eight meters and the sea level rises one to four meters (DELBOND ET AL. 2003).

Accumulation (Acc). One may not forget that besides erosion processes also accretion occurs along the Martinique coasts. The coastal system is highly dynamic: While the ordinary swell generally promotes the regeneration of the sandy beaches, the swell during hurricanes erodes them. On Martinique, data on beach recovery are lacking, but studies from other Caribbean islands show beach recovery rates of 74 to 99% in only four to seven months after a hurricane (CAMBERS 1996). Accumulation is therefore an important factor influencing the coastal shape but difficult to simulate in detail in a model. Nonetheless, we include the parameter “accumulation” in the hazard evaluation, because the sensitivity of the coast to erosion and inundation gets clearly accentuated through sedimentation processes. The intention is not to develop a high accuracy physical and hydrodynamic model of the coastal zone as has been done by e.g. CHEUNG ET AL. (2003). This is not possible due to the lack of accurate data. Instead, a generalized overview of potential erosion and accumulation sites shall be given. A rule-based approach is developed for the parameter “accumulation” that takes into consideration only relative statements of the size of the shelf area and currents as well as sediment supply of rivers. Currents mobilize substantial amounts of sediments. This drift also depends on the wind direction (KINSMAN 1965). Currents converge in the direction the wind blows (usually from east to west) and either push the sediment against the coast inducing sedimentation, or discharge it leading to the erosion of soft shores by an increasing offshore loss of sediment. On Martinique, the swell as sediment supplier is only of minor importance for the

coast. Often offshore loss of sediment takes place especially along the coasts with only small shelf areas. SAFFACHE (1999b) determines the origin of the main coastal sediments from inland area, transported to the littoral by rivers. CAMBERS (1996) also suggests that the shelf width is an important factor for beach recovery rates. Figure 1.2 gives an overview of this assessment. As result, this category yields information about the relative sediment supply and discharge of each single coastal part.

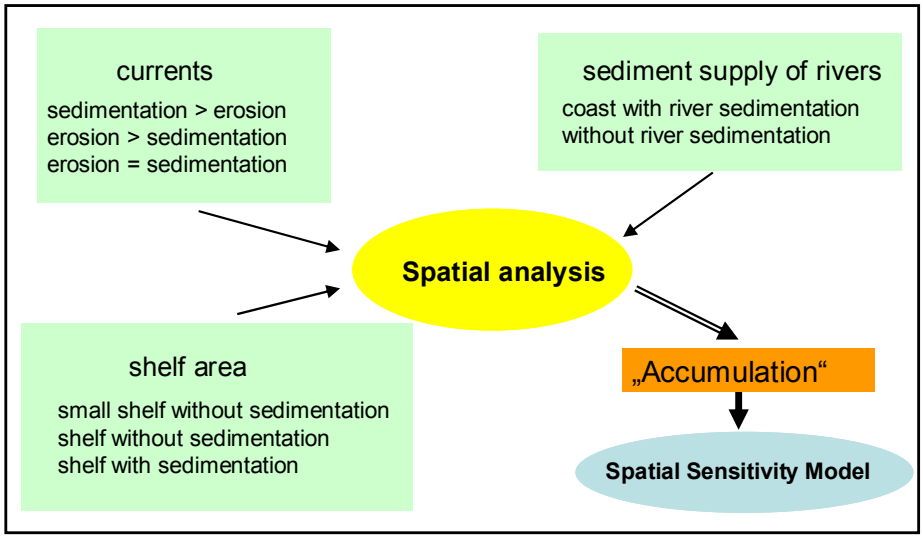


Fig 1.2 Overview of the "Accumulation"-model of sediment supply along the Martinique coast. The results serve as input for the sensitivity analysis.

The assessment of coastal sensitivity to hazards requires the objective integration of the different criteria into one model. In this study, the transformation of spatial information is performed by adding three fuzzy variables to each parameter ranging in weight from 3 ("high"), through 2 ("medium"), to 1 ("low") based on the correlation analysis. Table 1.1 exemplarily lists the vulnerability assignments and the index base for the categories "elevation" and "erodibility" (cf. Bacon 1994; Gornitz et al. 1997; CPACC 1999b).

Table 1.1 Vulnerability assignments for the parameter elevation and erodibility.

Sensitivity Index	**1 (low)**	**2 (medium)**	**3 (high)**
a. elevation	**high**	**intermediate**	**low lying**
after topography	> 20 m mountainous inland area	>10 to ≥ 20 m hilly inland area	0 - 10m flat land, lakes, wetlands
after morphology	steep coast lifted rocky shore	active cliffs low steep coast	sand and stone beaches rocky shore mangroves muddy bays
b. erodibility	**low**	**intermediate**	**high**
based on geology	volcano cones lava flows	lime unconsolidated volcanic breccia heat tuff	alluvium deeply weathered volcanites pumice tuff

This so called Coastal Sensitivity Index (CSI) is evaluated for the weighted extent to which the attributes are influencing coastal erosion and inundation. The CSI of the five parameters is the basis of the spatial model assessment that uses logical assignments adapted from Map Algebra. Tables 1.2 and 1.3 give an overview of the spatial rules of the sensitivity assessment for erosion and flood hazard of occurrence. The tables are read as follows: in table 1.2 the sensitivity to coastal erosion is high if the erodibility of the respective coast is ranked high and the accumulation is not high (top row), or if the erodibility is at an intermediate level, the protection status is open coastal, accumulation rates are low and the exposure is windward (second row). The same principle applies to table 1.3, where the sensitivity to flooding is low only if the elevation of the coast is high or medium (last row). The flooding sensitivity is high if the elevation is low and the protection status is open coastal (first row), or if the elevation is low, the protection status is partly sheltered, and the exposure is not leeward (second row). The combination of these rule-based statements results in a sensitivity map of the coastal zone of Martinique with spatial resolution of 10 m (see figure 1.4).

Table 1.2 Spatial rating scheme for evaluation of the coastal erosion vulnerability.

categories	erodibility			protection			accumulation			exposure		
Sensitivity	High	Intermedium	Low	sheltered	partly sheltered	open coastal	high	medium	low	leeward	windward	other coast
High	x	–	–	○	○	○	–	x	◙	○	○	○
	–	x	–	–	–	x	–	–	x	–	x	–
Medium	–	x	–	–	–	x	x	◙	–	○	○	○
	–	x	–	–	x	–	–	–	x	○	○	○
	–	x	–	x	–	–	x	–	–	–	x	◙
	x	–	–	○	○	○	x	–	–	○	○	○
	–	x	–	–	x	–	x	◙	–	–	x	◙
	–	x	–	x	–	–	–	x	◙	○	○	○
	–	x	–	–	–	x	–	–	x	x	–	◙
Low	–	x	–	x	◙	–	x	◙	–	x	–	–
	–	–	x	○	○	○	○	○	○	○	○	○

Legend:

x	condition that has to be fulfilled per category
◙	"or" = alternative to "x" in same category
–	"and not"
○	all included into rating

Table 1.3 Spatial rating scheme for evaluation of the coastal flooding vulnerability.

categories	elevation			protection			exposure		
Sensitivity	High	Intermedium	Low	sheltered	partly sheltered	open coastal	leeward	windward	other coast
High	–	–	x	–	–	x	○	○	○
	–	–	x	x	◙	–	–	x	◙
Medium	–	–	x	x	–	–	–	x	◙
	–	–	x	x	–	–	x	–	–
	–	–	x	–	x	◙	○	○	○
	–	–	x	–	x	–	○	○	○
Low	x	◙	–	○	○	○	○	○	○

1.2.2. Evaluation of the flood impact area

Besides coastal sensitivity to erosion and flooding, the potential extension of the impact area during coastal hazards is also evaluated. The hazard area can be determined by developing scenarios of different wave heights and wind speeds that imply different flood intensities. ZAHIBO ET AL. (2007) show that the surge height of cyclones increases with the wind speed. However, observed data reveal a great dispersion. The scheme of PARANAS-CARAYANNIS (1975) describes the

specific factors that may produce extreme water fluctuations at a coast during the passage of a hurricane. These factors are storm intensities, size, path, duration over the water, atmospheric pressure variation, tides, wave setup etc. The cumulative surge height results from frictional wind effects, atmospheric pressure changes, the phase of the astronomical tide, and the superimposed storm waves. Generally, the coastal exposure to a hurricane and the physical factors like protection of the coast vary. In the scenarios we initially assume the same conditions for each coastal segment irrespectively of their attributes. This has the advantage that the procedure is simplified and it becomes possible to concentrate on the storm intensities and the extent of the flood only. Four flood scenarios are created based on tropical storm and hurricane surge components and their resulting surge heights (PARANAS-CARAYANNIS 1975). If a hurricane of category 2 is passes through, the coastal areas are expected to become flooded by about 0.3 to 1.0 m. The first scenario assumes a tropical storm with 1 m hypothetical surge height, the second a hurricane of category 2 with 4 m, and the third scenario deals with a category 4 hurricane of 8 m surge height. The fourth flood scenario considers extreme events like tsunamis with 15 m surge height. During tsunami events wave heights might be higher than during hurricanes and the areas in danger of inundation often extends further inland due to the differences in wave physics (ZAHIBO & PELINOVSKY 2001). MERCADO-IRIZARRY & LIU (2004) give an overview of Caribbean Tsunami Hazards. According to LANDER ET AL. (2002), who compiled a Caribbean database of tsunami events, Martinique has been hit by at least five verified waves during the last 250 years. SAFFACHE ET AL. (2003) even assume seven waves during that time period. Therefore it is much more remarkable that even after the 2004 tsunami in South-East Asia no progress has been made in the evaluation of potential tsunami impact areas in the Caribbean. Most scientific studies focus on the generation, physics and detection of tsunamis (HELAL & MEHANNA 2008; NAJAFI-JILANI & ATAIE-ASHTIANI 2008) or concentrate on palaeogeographical events (FANETTI ET AL. 2008; SCHEFFERS ET AL. 2008). Others analyze the impacts of the 2004 tsunami in south-east Asia. COCHARD ET AL. (2008) give a review on coastal ecosystems, wave hazards, and vulnerabilities of this tsunami.

The four scenarios are coupled to a digital elevation model of the island (IGN 1996) to transform the scenario-specific water surface height into thematic layers of flooded and non-flooded areas. The compilation of all four layers results in an inundation map of the Martinique coast. COLBY ET AL. (2000) show that using DEM inundation modelling in a GIS leads to similar results as using a hydraulic model for evaluating the flood extent of hurricanes. Both models reasonably approximate the actual extent of flooding in that case study. The advantage of a DEM approach is not only its simplicity, but also the requirement of only stage level data (COLBY ET AL. 2000). However, the limitation is that rivers and their tributaries are only flooded according to the water height given by the scenarios and that the actual flows are not modelled. The results presented here show potential hazard areas of the island that assume equal physical conditions in each coastal segment. Through the sensitivity model described above the spatially explicit probability of erosion and inundation determined for each coastal segment can be compared with the extent of the flooded area.

1.2.3. Validation by recent and historical coastline changes on Martinique

To validate the model, we use recent spatial coastal data combined with historical data of flood extension (PERRET ET AL. 1996; SAFFACHE 1998; METEO-FRANCE 2000; SAFFACHE ET AL. 2002; CARIBBEAN HURRICANE NETWORK) and data on observed erosion rates (SAFFACHE ET AL. 1999; ASSAUPAMAR 2002). Information about historical tsunami damage in the Caribbean are used to validate its potential impact zone (HECK 1947; LANDER ET AL. 2002).

Coastal erosion is a major problem along the north-western shore of Martinique (SAFFACHE 1998). Within 40 years, on average 25 to 35 m coastline recession has been observed here (SAFFACHE & DESSE 1999). At Anse Belleville even more than 70 m of the coast eroded during that time (SAFFACHE 1998) and at Grand-Rivière community estimations state that marine erosion removed 50 m of land within 50 years (ASSAUMAMAR 2002). The reasons are natural as well as anthropogenic: The coast mostly consists of fragile material like unconsolidated volcanic rocks or alluvial sediments that are very sensitive to erosion. Even though the rivers transport enough sediment into the sea, the material that arrives on the north-western coast gets directly canalized and discharges quickly because

of the steep shore. Additionally, the swell is particularly erosive here, because no coral reef protects this part of the island from the swell. Also quarries along rivers in the northwest accentuate the erosion along the coast by hindering the supply of natural sediments to reach the beaches. In total, 200 000 to 350 000 m³ sand and gravel is extracted here annually (SAFFACHE 1998).

However, anthropogenic activities also lead to sedimentation along the coast: On the southern part of the island, rivers transport high sediment loads into the sea. From 1955 until 1994, the seaward progression at Marin and Galion Bay amounted to 30 m on average with a range between 15 and 70 m (SAFFACHE ET AL. 1999; SAFFACHE 2000). This progression is caused by the denudation and the erosion of the soil from agriculturally used watershed areas because of the intensification of banana cultivation in combination with higher soil erosion rates. Accumulation of sediments along the coast is generally facilitated in shallow waters. But the sediments originating from land are often polluted with high concentrations of fertilizers and other chemicals brought onto the fields. The polluted sediments threaten the most relevant habitats along the coast and inside the bays, such as mangroves, coral reefs, and seagrass beds. There are plans to reduce, respectively to entirely curb this hyper-sedimentation in the bays by improved land use practices (SAFFACHE ET AL. 1999; PUJOS ET AL. 2000). Based on historical storm surge heights on Martinique spatial analysis reveals that elevations below 2 m, 5 m, and 10 m show the greatest probability to get flooded depending on the intensity of the storms. Additionally, at the north-western coast at l'Anse Bellevue, the erosion rates during hurricanes have been extremely high in the past. SAFFACHE (1998) computes retreats from 2.5 to 7 m per cyclone event.

Analyses reveal, that the spatial sensitivity model indicates high sensitivity to flood and erosion where historical and actual erosion and inundation have been observed. Also, the scenario-derived development of inundation area approximates the historical extents of flooding. These results seem trivial but at the same time they highlight the accuracy of the model. This not only allows realistic statements of actual coastal sensitivity but also the additional development of sea level rise or other coastal change scenarios in future studies of the island.

1.3. Results

The coast of Martinique consists of four main coastal types that can be defined as sandy bays, muddy bays, rocky shores, and steep coasts (cf. figure 1.6). In the north of the island the coastline is steep and smooth, whereas in the southern part it is flatter and disturbed by numerous bays, islands and peninsulas. The low lying coastal areas dominate with about ¾ of the total coastal expansion (~ 326 km), of which the rocky shores derive by far the longest total coastal extent with nearly 180 km (42%), followed by 79 km (18%) of mangrove forests. The mangroves can be found in the Ford-de-France bay and on the southern and eastern coast. Whereas sandy beaches (13%) and rocky shores are distributed over the entire coast, active cliffs (5%) are only found at the northern and western coast of Martinique. Considering only the geology, 82.3% of the coastline's sediments and rocks are highly erosive. On Martinique, alluvial material is most common (163 km), followed by deeply weathered vulcanite (119 km) and unconsolidated tuffs (65 km). The category "natural protection" on the other hand shows that 34% of the Martinique coastline is protected naturally, 10% is only partly sheltered, while the majority of 56% of the total shoreline is attributed as "open coastal".

The sensitivity study reveals that only 11% of the total coastline of 432 km has a low sensitivity to erosion, while the majority is either medium sensitive (48%) or has a high sensitivity (41%). The northern coastline between Fort-de-France and the Caravelle peninsular is particularly endangered. This section is characterized mainly by loose, unconsolidated material. The most sensitive rock types to erosion during hurricanes are the pumice formations in the northern part of the island. But also along the eastern coast, sections with high erosion possibility can be found. These are mainly the coastal parts of peninsulas towering into the sea. The bays between them on the other hand are less prone to erosion. The few coastal sections with very low erosion impact are found in the south-western island close to Les Anses d'Arlet but also along the Atlantic coast, for example at the southern Caravelle peninsula. The accruing rock formations here are marked with low erodibility.

The evaluation of flooding sensitivity reveals that more than one fourth (27%) of the total coastline has a low sensitivity to hurricane flooding. These are, as

anticipated, the coastal parts that are characterized by steep coasts with heights over the expected flooding mark. Nevertheless, 28% of the entire coast has a high probability to get flooded during storms. These endangered areas can be found at the northern island between Fort-de-France and La Trinité as well as along the coasts of the greater peninsulas Caravelle, Les Anses d'Arlet, or Sainte Anne. But especially along the northern coast only very narrow coastal strips are involved, with quickly rising mountainsides in the hinterland. Altogether, 45% of the coastline has at least a medium vulnerability to flooding and inundation. Many bays along the southern island and also the Fort-de-France bay are rated with medium sensitivity. Coral reefs are mostly located in front of the bays, which are reducing the wave pressure, or islands and peninsulas protect the coast in the main storm direction. The coastal segments with a medium sensitivity to flooding own a small probability to get inundated during hurricanes, but fall into the impact area of more extreme events like tsunamis.

In summary, 13 % of the total coast of Martinique show a low sensitivity, 43% have medium sensitivity, and 44 % are highly vulnerable of coastal flooding and erosion. The relative distribution of sensitivity in relation to erosion and flooding is represented by the two-digit CSI. Figure 1.3 illustrates the relative total length of each CSI.

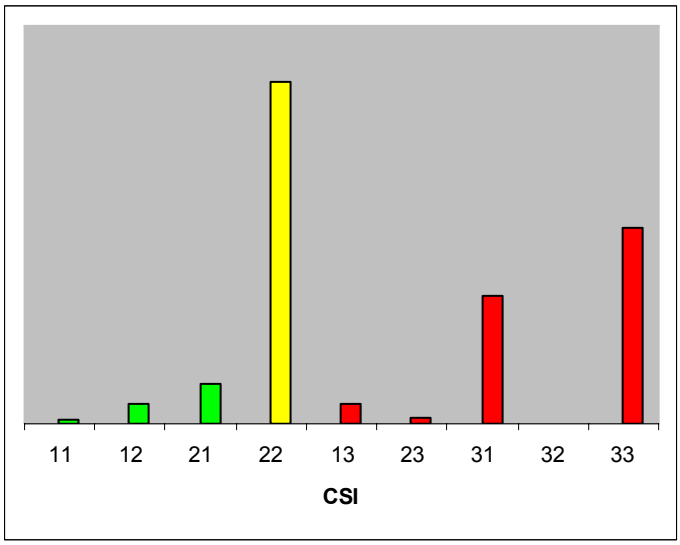


Fig 1.3 Relative coastal length of sensitivity in relation to erosion and flooding.

The first digit describes the erosion possibility from low (1) to high (3), and the second digit of the CSI denotes the flooding possibility in the same way. The CSI

that covers at least one „3“ is categorized as „high total sensitivity“, and the CSI that contains at least one „1“ (and no „3“) is rated as „low total sensitivity“. “Medium sensitive” are those coastal parts with a CSI of “22”. For example, a CSI of “23” means medium erosion possibility (2) but high flooding possibility (3) of the selected coastal area. It is rated as “high” total sensitivity. The analysis shows that the CSI of „22“occupies by far the greatest length of the entire coastline followed by the CSI of “33”. Figure 1.4 finally illustrates the results of potential erosion and flooding in a map.

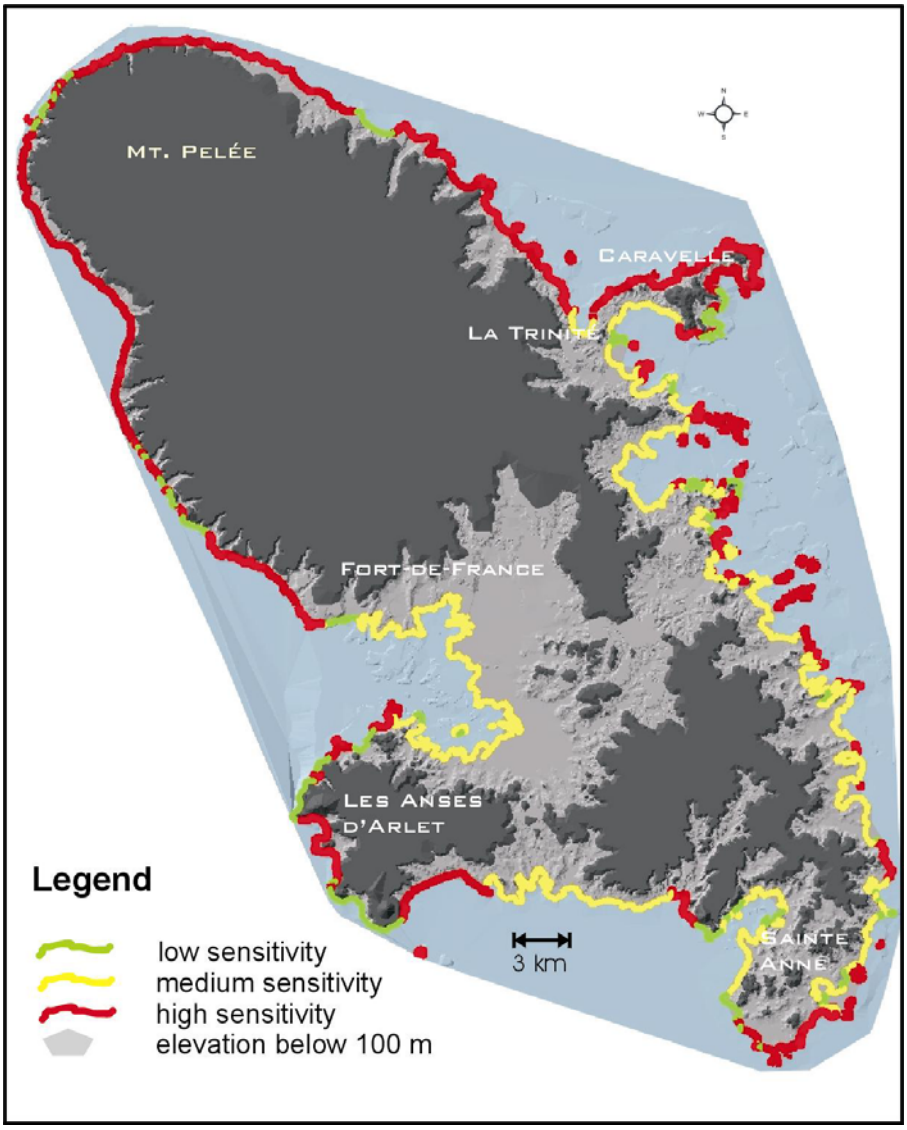


Fig 1.4 Map of the coastal sensitivity to erosion and flooding on Martinique.

The map shows that the coastal parts with low sensitivity are well distributed over the whole island but only in small coastal areas. The coastal segments least vulnerable are attributed with “lifted rocky shore” consisting of hard rock lava

flows in leeward exposure along the southwest coast at Anses d'Arlet. Coastal areas with medium sensitivity are only found along the southern island coasts mainly inside the bays and cays whereas the towering coastlines in between are rated as highly sensitive, just as most of the coastal parts along the northern island. To sum up, the coastlines most vulnerable to flooding during hurricanes are situated along the northern island between Fort-de-France and La Trinité as well as along the coasts of the greater peninsulas Caravelle, Les Anses d'Arlet, or Sainte Anne.

The hazard area map shown gives an overview of the extent of the flood hazard zone. For the evaluation of the impact area several scenarios are developed. Spatial analysis identifies the areas that are likely to be affected by floods. Figure 1.5 shows the results. The high impact areas are highlighted in black and dark grey on the map. Elevations above 100 m are not explicitly shown, because they are insignificant for this analysis. Analyses show that 58 km² have a very high flood possibility from any coastal hazard including tropical storms, 55 km² lie in the range of a high impact possibility at moderate hurricanes, and 57 km² have medium impact possibility. Altogether, this amounts an area of 170 km² or about 16 % of the islands surface. The greatest expansion of the coastal impact areas can be found in the Fort-de-France bay and at the bays of the south-western island. In contrast, in the northern part of the island, which is dominated by the volcano Mt. Pelée, merely small, narrow areas show sensitivity to flooding. But this small land adjacent to the beach is often the most valuable for the local communities. Here, the vulnerability to erosion and inundation is also very high as the sensitivity model shows. In the interpretation of the maps it is important to consider both maps: the sensitivity map to obtain statements about the probability of becoming impacted by coastal hazards and the hazard area extent map that yields information about the potential impact area, if a certain hazard occurs.

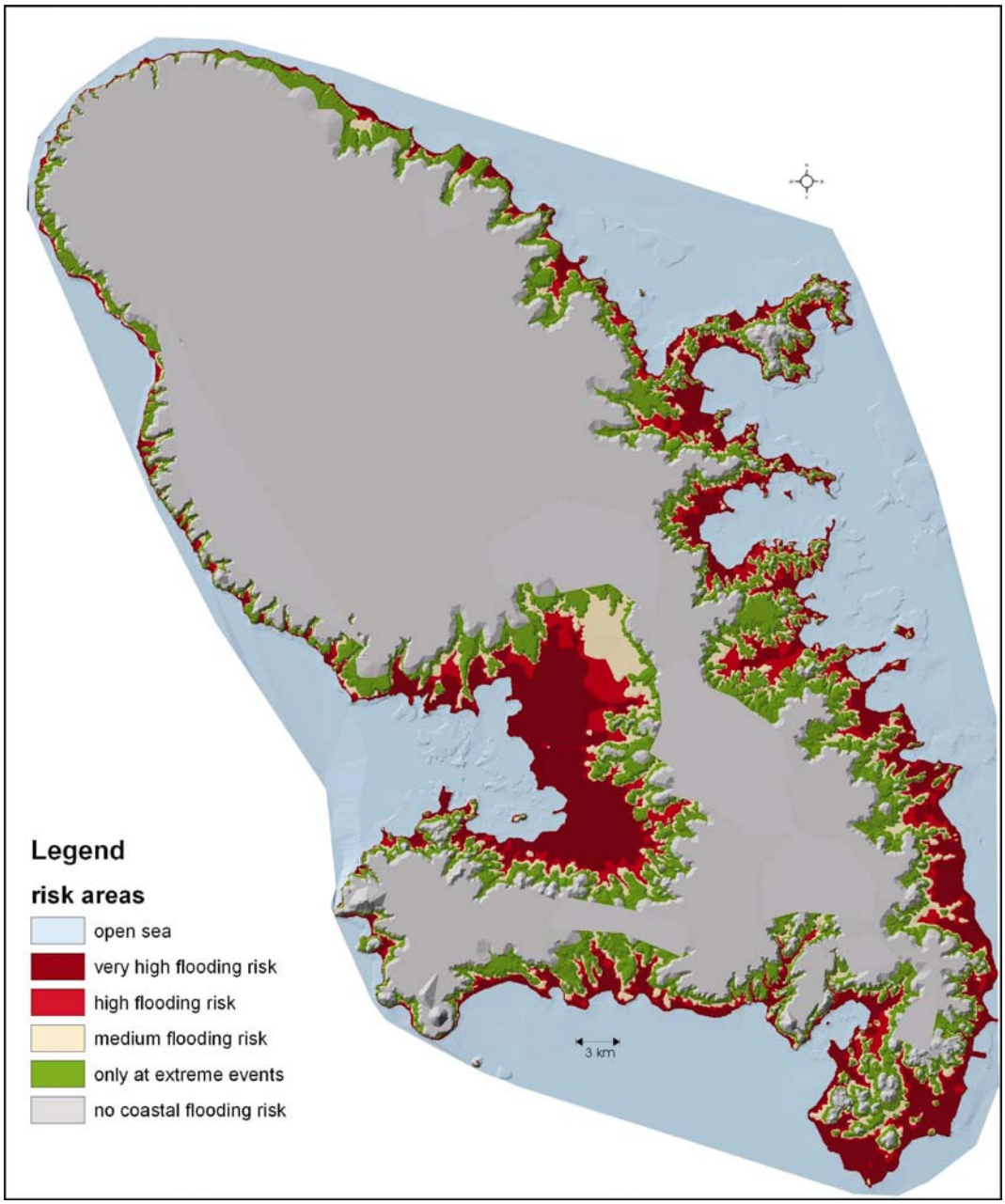


Fig 1.5 Map of the hazerd areas to flooding during hurricanes or other extreme coastal flooding events like tsunamis.

1.4. Discussion and Conclusion

Tropical cyclones and hurricanes are frequent in the Caribbean region. On average, every 3.6 years a cyclonic phenomenon passes in proximity of Martinique; the latest one was hurricane Dean in August 2007 that caused severe damage on the island. With a continuation of global warming, more frequent and intense hurricane activity is expected (PARRY ET AL. 2007). This highlights the importance to be aware of potential local hazards induced by these hurricanes.

This study concentrates on the impacts of coastal erosion and inundation caused by coastal hazards using the Lesser Antilles island of Martinique as case study site. Most of the settlements on the mountainous island are situated along the coast and coastal tourism is one of the main income sources. Often sensitivity assessments require detailed and complete data. This limits the applicability at coasts with inaccurate data structure. The methodology presented here relies on empirically and rule-based statements that are combined logically in a GIS-environment. This way it was possible to locate coastal areas that are highly sensitive to the consequences of coastal hazards. The analysis shows that nearly half of the Martinique coast is vulnerable to flooding and erosion. The most sensitive coastlines are those along the northern coast and also the southern coasts that tower into the sea. The extent of the flood hazard zone is determined through the combination of scenarios and a DEM. This methodology seems simple, but produces similar results as a hydraulic model (COLBY ET AL. 2000). The advantage is that this method requires stage level data only, but on the other hand additional river floods are not considered in this analysis. The cyclonic variance in wind speed and wave forces is described by the flood heights of the scenarios. Therefore, the analysis does not compete with more detailed physical and hydrodynamic models as conducted by e.g. Cheung et al. (CHEUNG ET AL. 2003). However, it gives planners a generalized overview of potential impact areas of erosion and inundation. The defined surge heights are only hypothetical, but the scenarios may be coupled with hydrodynamic models of hurricanes to obtain more realistic flood heights for particular hurricanes. The validation with historical and actual inundation data and erosion rates showed high accuracy. As a consequence studies about the impact of changes in sea level or hurricane intensity are conductible through this spatial model as well.

The methodology is easy applicable and can be projected to other islands or coastlines as well. In addition, the information gained from the spatial analyses is useful as basis for the conduction of more detailed local studies. The information derived from spatial analyses is also useful for everybody interested in determining the present and future vulnerabilities of coastal zones to erosion and inundation if data from hands-on measurements are scarce or not readily available. This includes the evaluation of damage costs caused by hurricane or

tsunami flooding. The results obtained from the GIS-based model fill the gap of missing data sources.

1.5. Annex

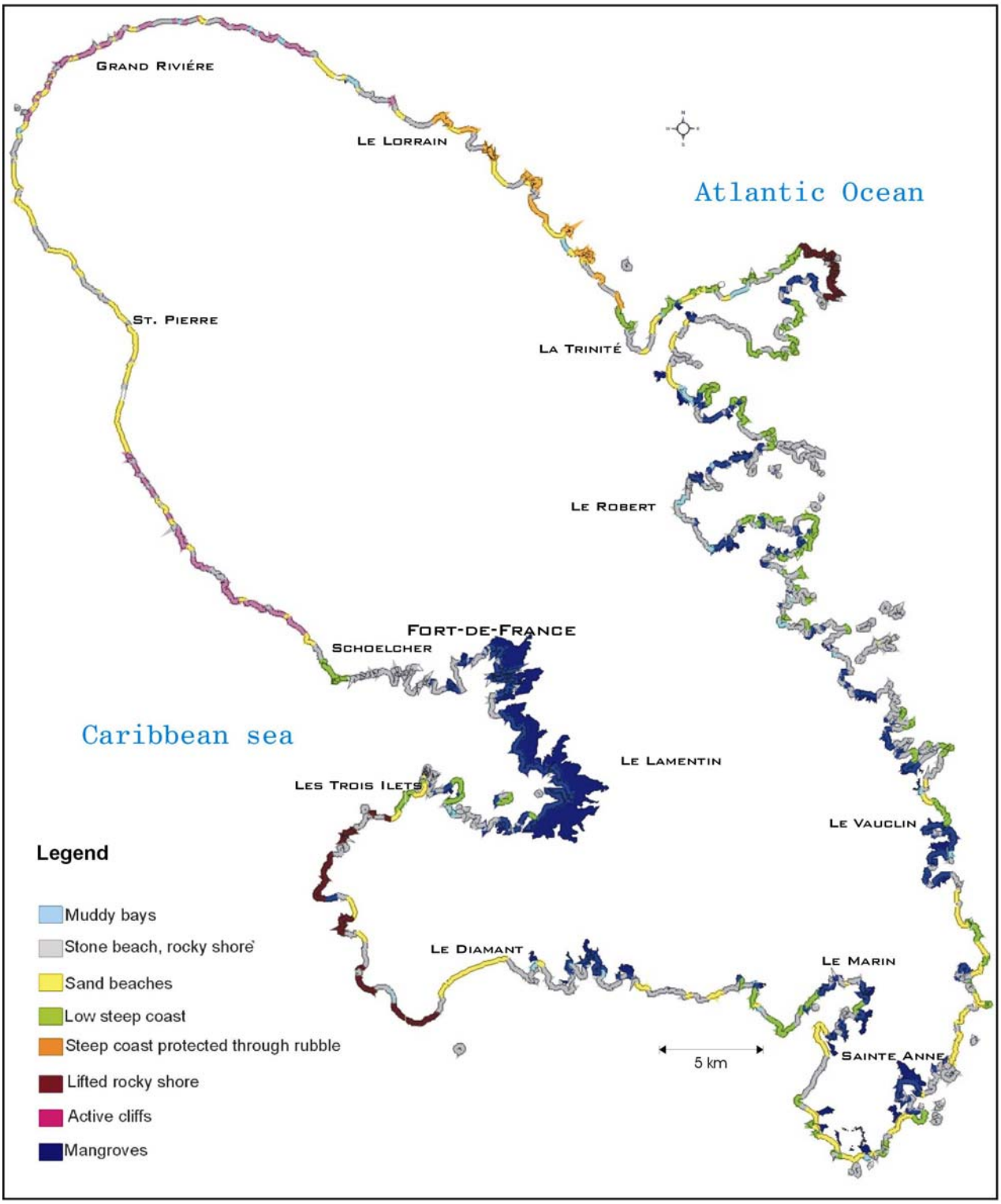


Fig 1.6 The distribution of coastal types on Martinique.

2. Evaluation of coastal squeeze and its consequences for the Caribbean island Martinique

2.1. Introduction

2.1.1. Coastal squeeze

The likely responses of wetlands to sea level rise are: Loss of the total wetland area by coastal erosion and inundation, relocation or migration rather than overall loss, change in the mangrove forest or beach structure, and mangrove increase or new beach accretion further inland. The aim of this study is to develop a methodology to evaluate the impacts of accelerated sea level rise on the beaches and mangrove wetlands of Martinique.

As described below, these wetlands are of special importance for the island's economy and ecology. This study especially considers the sensitivity of the wetlands to coastal squeeze. In general, the term "coastal squeeze" is applied to the situation where the coastal margin is squeezed between the fixed landward boundary (artificial or natural) and the rising sea level (ENGLISH NATURE). Most studies analyse squeeze in combination with tidal habitats (DOODY 1992; LEE 2001; SALMAN ET AL. 2004), but the term can also be used for other habitats when the landward position is fixed and erosion at the seaward margin is taking place. Numerous studies to the impacts of accelerated sea level rise exist, though only a few especially address the coastal squeeze problem (JONES 2001; DOODY 2004). Figure 2.1 shows a scheme representing the factors that influence coastal squeeze.

2.1.2. Sea level rise in the Caribbean

Relative sea level in the Caribbean has risen by about 20 cm within the last century (MAUL 1993). Regional projections state a rise in sea level of additionally 10 to 50 cm by 2025 (MAUL 1993; UNEP 2000), respectively approximately 65 cm by 2100 (IPCC 2001). Scarce data availability within the Caribbean and high spatial variability among the islands makes concrete relative sea level rise estimates for each single island region problematic. Besides a rise in sea level further projections for the Caribbean region (UNEP 2000) expect an increasing

frequency and intensity of hurricanes and tropical storms that coincide with coastal flooding and high erosion rates at the shores, also as a cause of rising sea level.

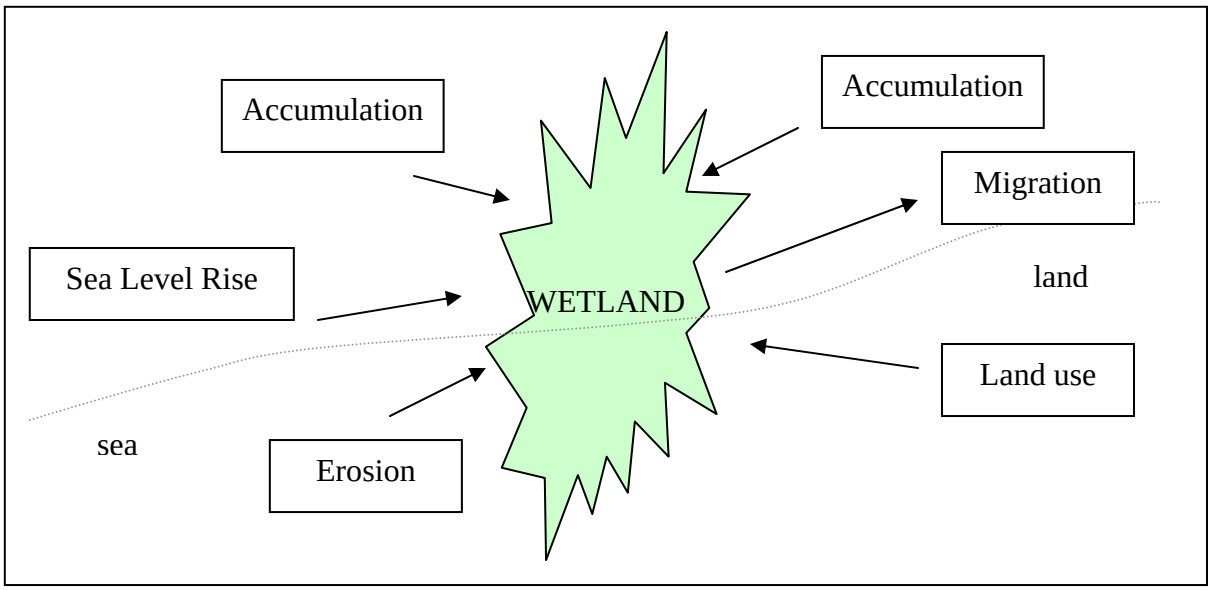


Fig 2.1 Scheme of factors influencing coastal squeeze.

2.1.3. Mangrove response to climate change and sea level rise in the Caribbean

Several studies about likely wetland responses to sea level changes have been conducted (e.g. ELLISON 1993; UNEP 1993; VAN DAM ET AL. 2001; IPCC 2002). However, there is still much controversy about this subject. Scenario studies of mangrove responses to sea level rise vary from little adverse impact to collapse. General statements should be treated with caution and some authors demand site-specific analyses (BACON 1994). The conclusions from these studies can be summarised as follows: The health of mangrove forests is mainly influenced by sediment supply/flux, suitable substrate, stand composition and status, tidal range and migration opportunities (VAN DAM ET AL. 2001). For this reason, the impact of sea level rise on coastal ecosystems will vary regionally and will depend on erosion processes from the sea and depositional processes from land (IPCC 2002). As the sea level rises, the surface of a coastal wetland shows increased vertical accretion due to increased sediment and organic matter input (NICHOLLS ET AL. 1999). Therefore wetlands show a dynamic and non-linear response to sea level rise. Studies of the UNEP (1993) expect mangrove forests to tolerate the

anticipated sea level rise in rainfed humid areas. But the mangroves may be overstepped and abandoned in more arid areas particularly if inland retreat is not possible (UNEP 1993). Fringe mangroves, the main mangrove type on Martinique, are expected to decrease in area on mountainous islands, and to migrate inland on low lying islands (BACON 1993). River mangroves are projected to be able to migrate inland. But, in many cases on Martinique, coastal development prevents wetlands from adapting by shifting landward. Where wetlands are bounded by elevations, as is the case in many of the Caribbean islands, it is unlikely that they will shift landward as the sea level rises (ELLISON & STODDART 1991; UNEP 1993). Accretion studies of ELLISON (1993) show that mangroves of low islands are at risk from the rates of sea level rise predicted for the next 50 years. They are expected to suffer from erosion and inundation stress. Studies for mountainous islands, such as Martinique, do not exist.

2.1.4. Martinique

Martinique is an island of the Lesser Antilles in the Caribbean region. It is a French Department and EU "ultra-peripheral region". The economy is largely based on the export of agricultural goods (bananas, sugarcane, and pineapples) and tourism as major income sources. Nearly one million visitors annually arrive on the island that is inhabited by approximately 390.000 people (MARQUES 2002; CHARRIER 2003). Because the topography of the island is characterised by steep mountains, the majority of the settlements and about 77% of the population are situated along the coast below the 20 m contour line. Today, most of the Martinique population is concentrated in the extending urbanized zone of the cities Fort-de-France and Schœlcher, where houses are built almost at the level of the sea.

2.1.5. Beaches and mangroves on Martinique

Figure 2.2 shows the distribution of beaches and mangroves on the Martinique coastline. Martinique has approximately 120 beaches which make up 13% or 57 km of the entire coastline. The beaches on the north and south coasts of Martinique are made of fine sands. The northern beaches are situated between the foothills of the Mt. Pelée and the sea. They mainly consist of black sand that originates from the erosion of pyroclastic depositions of volcanic eruptions in the

hinterland. In general these beaches are small in extent. The southern beaches consist of white sands that originate from the abrasion of bordering coral reefs along the Atlantic. They are of special importance for tourist purposes. The main endangerment for beaches is erosion. Currently eroding beaches and barriers are expected to erode further as the climate changes and sea level rises (IPCC 2002).

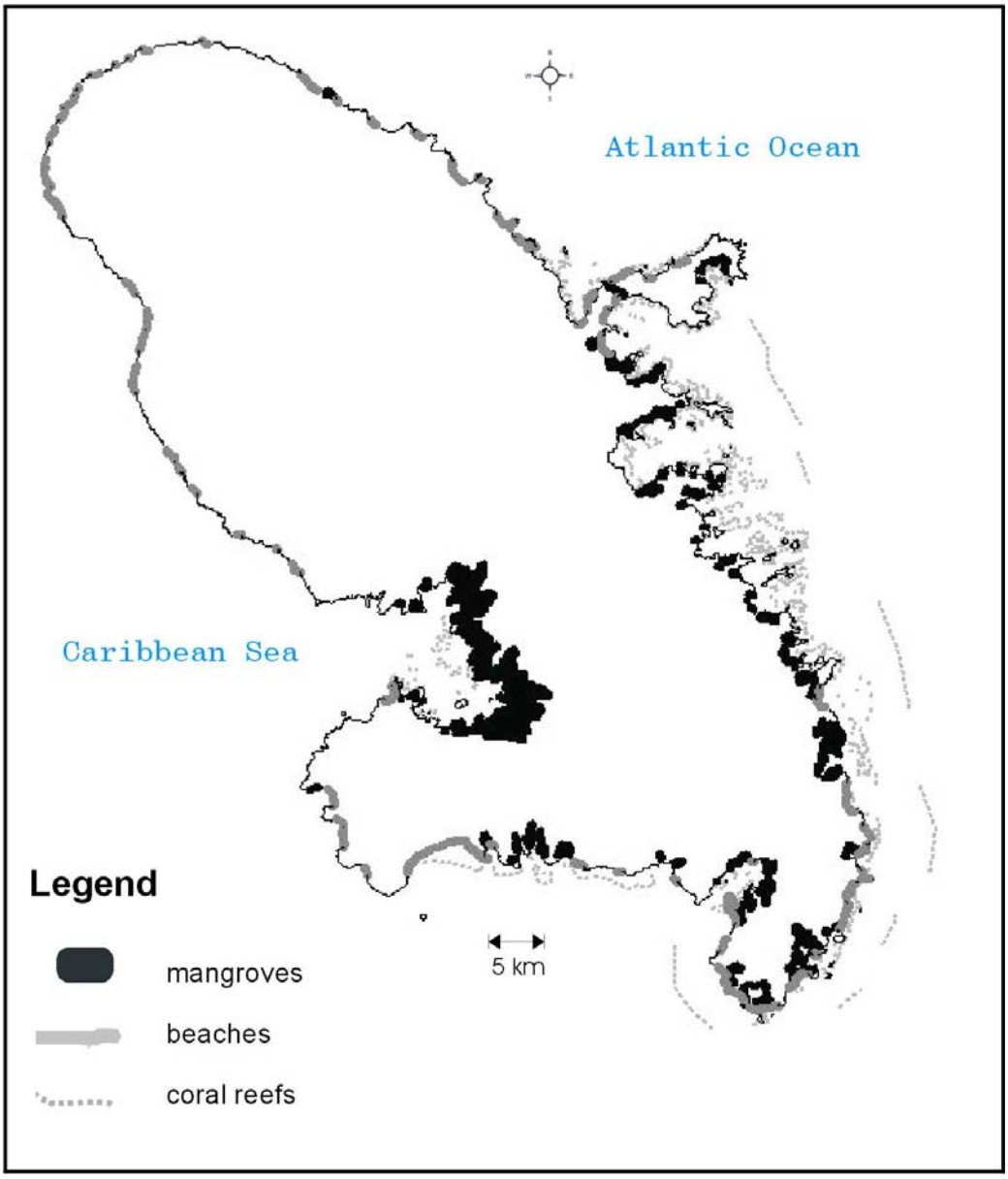


Fig 2.2 Distribution of beaches and mangroves on Martinique (based on IGN 1996; HINNEWINKEL & PETIT 1975).

Along the Martinique coastline there are about 79 km (18%) of mangrove forests, which total approximately 1 850 ha of mangrove area or 6% of the total island surface (DAF-AGRESTE 1998). Mangroves are mainly found inside the bays of the south and south-eastern coast of the island (650 ha), as well as in extensive formations in Fort-de-France Bay (1 200 ha). On Martinique one can distinguish two mangrove types (DELBOND ET AL. 2003): First of all there are mangrove littorals or submerged forests with *Rhizophora sp*. These types are situated inside the bays, mainly at Fort-de-France Bay, Francois, Robert, and Marin. Secondly, there are mangroves bordering the river mouths (Brossard et al. 1991). These types can only be found at Trinité and growths on terrestrial sediments. Mangroves serve as main nursery areas to commercially important fish stocks. They are also home for a large variety of birds, reptiles and mammals. The wetlands at Fort-de-France Bay are internationally important for migratory birds (UNEP 1989), and the Martinique mangroves are rich in molluscs and crabs. Despite its (biological) value (see also COSTANZA ET AL. 1997), Martinique mangrove swamps are often regarded as marginal land and are therefore systematically degraded and destroyed (GABRIE ET AL. 2004). Often they have been selected as dump sites or for housing or other urban structures to accommodate coastal development (LEWSEY ET AL. 2004). In the past, mangroves have been cleared especially for the development with tourism-related and residential buildings, due to urbanisation pressure, unsustainable wood use, or industrial pollution. GABRIE ET AL. (2004) report that about 30 % of the mangroves on Martinique were lost between 1972 and 1992. Even though there are now measures to prevent mangrove destruction which also include the construction of purification plants and the treatment of industrial sewage at an European norm, most of the mangroves still suffer from degradation (IFRECOR; ASSAUPAMAR 2002) and loss (FAO) (see figure 2.3).

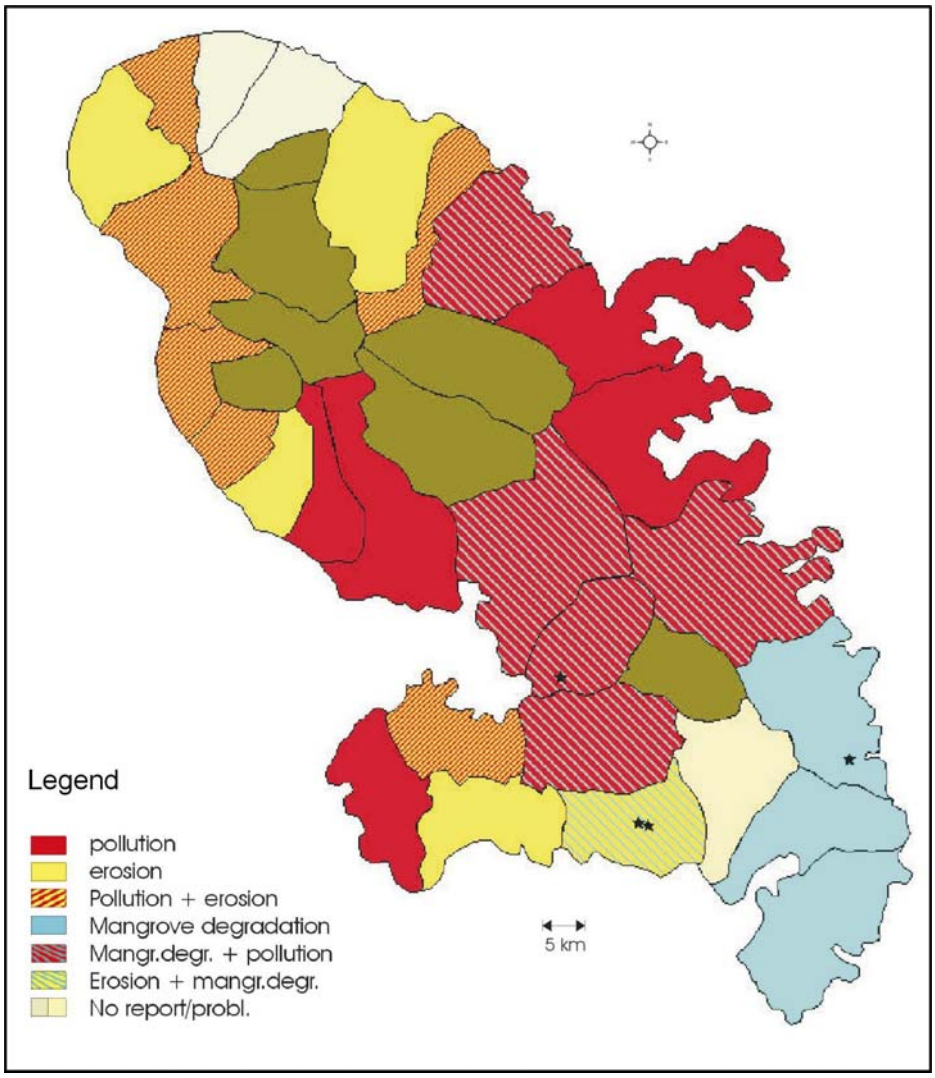


Fig 2.3 Reported environmental problems of the coastal zone per district. Data adapted from ASSAUPAMAR 2002.

2.1.6. Tourism on Martinique

Tourism is, besides the export of agricultural goods, the most important income source on Martinique providing 6.4% of the GDP (CARIBBEAN TOURISM ORGANIZATION 2004). Especially the beaches along the Martinique south coast are famous among tourists. Here the majority of tourist accommodation facilities, mainly bigger hotel complexes are situated because of the favourite climatic and coastal conditions. Every year thousands of tourists visit the island just to spend some time (14 days on average) on these beaches. Main tourist season is during the dry months between December and March. From 1960 to 1998 the number of visitors has multiplied by about 50 (OFFICE DÉPARTEMENTAL DU TOURISME). Most visitors come to Martinique for its “sun, sea, surf and sand” image. More

than three quarters of the visitors are of French origin (2003: 79%), whereas most of them come from the Départements d'Outre-Mer - Territoires d'Outre-Mer (DOM-TOM) (MARQUES 2002).

2.2. Methodology: evaluation of coastal squeeze and its consequences

The methodology is divided into two main parts: the first one is the evaluation of the coastal squeeze risk of beaches, mangrove forests, deltaic and estuarine areas, as well as coastal swamps resulting in a map and the second one is the detection and location of the most vulnerable tourist destinations to squeeze impacts on Martinique. Figure 2.4 illustrates the applied methodology in more detail. Additionally, table 2.1 gives an overview of the used data and its sources to examine the risk of coastal squeeze and area reduction.

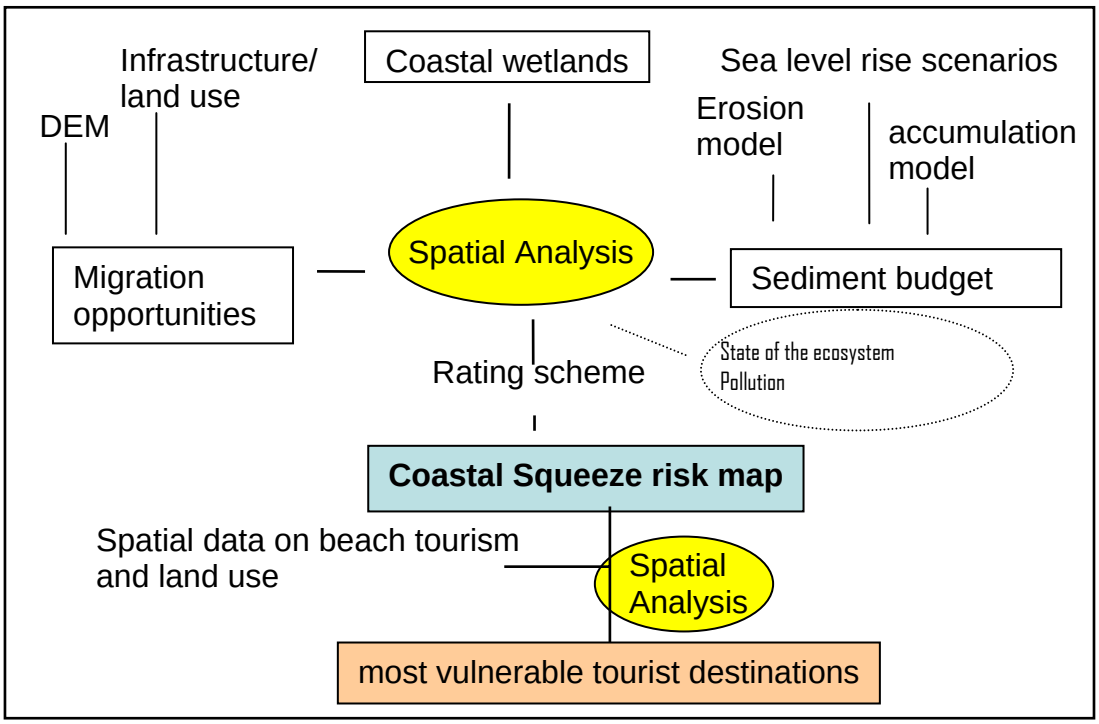


Fig 2.4 Overview of the methodology.

Table 2.1 Data and their sources for the evaluation of coastal squeeze.

Data	Sources
coastal elevation	HINNEWINKEL ET AL. 1975, IGN 1996, LANDSAT ETM+ SATELLITE DATA
coastal sensitivity to erosion and inundation during hurricanes	SCHLEUPNER 2007
land use	IGN 1996, LANDSAT ETM+ SATELLITE DATA,
wetland distribution	IGN 1996, LANDSAT ETM+ SATELLITE DATA,
sedimentation	ELLISON 1993, IFREMER.FR, SAFFACHE ET AL. 1999, SAFFACHE 2000

To evaluate coastal squeeze a spatial GIS-based model has been developed. Two parameters are selected that reflect the sensitivity of beaches and coastal wetlands to coastal squeeze and area reduction most: migration opportunities inland and sediment budget. These parameters were added to the spatial model that also gives geo-information about the coastal wetlands. The parameter "migration opportunities" rely on the variables human infrastructure and land use and a digital elevation model of which the inclination has been computed. The parameter sediment budget on the other hand refers to sea level rise scenarios of 25, 50 and 100 cm (IPCC 2001). It is conducted in an erosion model and an accumulation model. The erosion model is based on historical erosion rates as well as results from a sensitivity evaluation (SCHLEUPNER 2007). In addition, the Bruun-Rule has been applied to the spatial model (BRUUN 1962). "Accumulation" as a weighted spatial submodel on the other hand takes the size of the shelf area, currents as well as sediment supply of rivers into consideration. Due to lack of detailed information data from studies that determine accretion rates for mangrove swamps on other Caribbean islands are used. Such rates seem to be similar on different Caribbean islands. This makes it possible to utilise the data also for Martinique even if there is a high range of uncertainty because of its mountainous topography. For example, the accumulation rates from Bermuda, Tonga and Cayman average from 8 to 10.6 cm/century (ELLISON 1993). That is less than the rate of sea level rise of 14.3 cm/century over the last few centuries, and the average rate of expected sea level rise of 28 cm/century during this century. Even if the hinterland is steep or developed there may be no wetland change at all due to high accretion rates. SAFFACHE (1999a) examined sedimentation rates in bays of Martinique and found that hyper-sedimentation takes place that leads to enormous accretion rates inside the bays. Measurements

of the Direction Départementale de l'Equipement reveal that the river Lézarde deposits on average 100 000 m³ of sediments into the Fort-de-France Bay every year, the river Monsieur 45 000 m³ and the river Salée 90 000 m³ (cit. in SAFFACHE 1999b). The reasons are high erosion rates in the intensified used watershed areas. Due to improved management techniques the erosion and with it the hyper-sedimentation in the bays is going to be minimized to normal level in the forthcoming years. Additionally, these sediments are often polluted and vegetation stocks are stunted because of this.

The squeeze model also contains the parameter "state of the ecosystem". At the moment there is a lack of detailed information to this. Therefore, the state of the ecosystem has not been included into the model so far but it is highly recommended to do this if data are available. This problem will be discussed in combination with the results. For evaluation of coastal squeeze the parameters migration opportunity and sediment budget were analysed within the GIS-based model with help of a rating scheme that is described in more detail below. This scheme has been adapted from BACON (1994) and altered.

1. migration opportunities depending on morphology and development **(M)**
 a. landward margin steep or coastline developed = **High**; migration impossible **(3)**
 b. landward margin of reclaimed land = **Medium**; migration under restriction possible **(2)**
 c. landward margin an adjoining wetland = **Low**; migration possible **(1)**

2. sediment budget **(S)**
a. sedimentation < sea level rise = **High (3)**
b. sedimentation and erosion offset = **Medium (2)**
c. sedimentation > sea level rise = **Low Sensitivity (1)**

The results of the squeeze model are illustrated in a map that serves as base for the second part of this study, namely the detection of the tourist places most vulnerable to squeeze. Beaches are the main tourist destinations on Martinique. And also the mangrove forests serve as visiting places for tourists giving them the imagination of an adventurous day trip into "untouched nature". Spatial data on beach tourism (DAF-AGRESTE 1998; MARQUES 2002; PARA ET AL. 2002; MARQUES 2003; UYARRA ET AL. 2005; ARDTM; ESPACES; INSEE; OFFICE

DÉPARTEMENTAL DU TOURISME) and infrastructure/land use were combined with the results of the squeeze risk analysis. Spatial analysis is carried out that evaluates the tourist destinations most vulnerable to the impacts of coastal squeeze.

2.3. Results

2.3.1. Sensitivity of Martinique wetlands and beaches to coastal squeeze

In many areas of Martinique, beaches, mangroves and other wetlands provide significant coastal protection (DELBOND ET AL. 2003). The ecosystems serve as natural shock absorbers and protect coastal infrastructure and land use against tropical storms and hurricanes. They also provide critical storage capacities for storm surges and flood waters (COSTANZA 1997). Beaches serve as attractions for tourists and are of economic value. All together, in 2002 6 537 people were involved in the tourist business on Martinique, 4 534 of them are directly employed in hotels and restaurants, 1 390 indirectly through other services, agro-industry, agriculture or transport, for example (PARA ET AL. 2002). Loss of the coastal systems would therefore have great impacts on human life on Martinique. The following analysis examines the adaptive capacity of wetlands and beaches to sea level rise on Martinique. Altogether, 78% of the mangroves, 98% of all Martinique beaches, and 86% of other coastal wetlands are at risk to erosion and inundation if sea level rises. Often inland migration is impossible not only because of topographical reasons but also because of urbanisation. Even if the majority of the coastline is natural space, 32% (916 ha) of the coastal zone, the so called “50 pas”, are urbanised. Figure 2.5 visualises the sensitivity of coastal segments to area reduction and coastal squeeze for a sea level rise of 25 cm. About 45.6% of the coastline is very sensitive to coastal squeeze.

The endangered segments are distributed all over the island, and especially the north-western part of the island is at high risk. Here the anthropogenic developments are situated right between the sea and the steep slopes of Mt. Pelée. Without their high sedimentation rates bays at the southern and eastern coast and the Fort-de-France Bay would also have greater sensitivity indexes than evaluated. While 28.8% (=124.6 km) have a medium squeeze index, only 0.8%

(3.4 km) got the “low sensitivity” index. 24.8% (107.5 km) have not been included in the analysis because of their morphology (e.g. steep coast etc.).

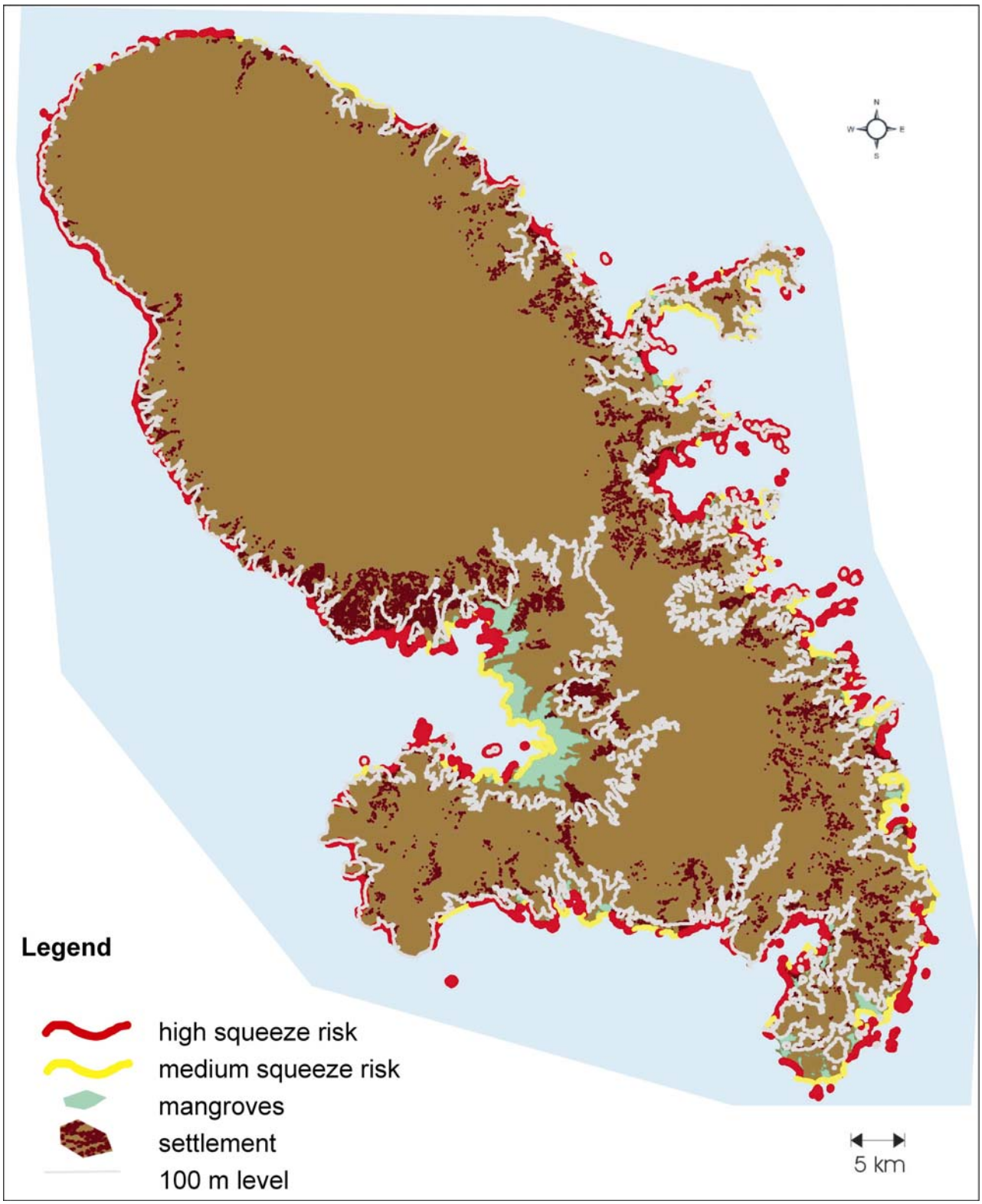


Fig 2.5 Sensitivity of the Martinique coast to area reduction and coastal squeeze at accelerated sea level rise.

2.3.2. Human impacts on mangrove squeeze and beach reduction

On Martinique tourism infrastructure, road networks and major settlements are usually all located along the coast giving locals and visitors an easy access to the coastal and marine natural resources and hindering the wetlands from migrating further inland. The vulnerability of beach reduction is also augmented. An evaluation of infrastructure and constructions situated within the impact zone reveals that settlements are seldom found below an elevation of 5 m along the southern coast whereas at the northern coast they reach further down to sea level because of settling limitations of the mountainous hinterland. On average, the majority of coastal constructions are built at heights between 5 and 10 m above the present sea level and therefore also within the zone at risk of flooding and erosion. Especially the tourist hotels are mainly found very close to the sea and below the 5 m level. Additionally, many coastal districts in the south experienced massive population increases during the last decade: Le Diamant +40%, Trois Ilets +38%, Sainte-Luce +30%, Rivière-Salée +30%, and Saint-Joseph +25%, for example (CONSEIL REGIONAL). Figure 2.6 shows the districts with more than 50% coastal urbanisation as well as the population evolution per district over time. Especially in the north-western coastal part urbanisation rates are quite high because of the mountainous topography. But southern districts also, at the moment considered as the most attractive living space, show high urbanisation rates.

Massive hotel construction programmes led also to a profound transformation of the coastal zone. In the past, the majority of tourism development has taken place without prior environmental assessments. As a result, hotels have been constructed in areas of valuable natural habitat. Latest trends have even led towards high-density, mass-market tourism sites close to the water's edge. But also smaller holiday and weekend homes are found along the south and south-eastern coast mainly between Trois-Ilets and Trinité. It is therefore a quite logical consequence that the majority of southern districts experience mangrove degradation (cf. figure 2.3). This may accentuate the coastal squeeze and would give way to the sea to reach further inland than with land protecting mangrove forests. Also coastal pollution adversely affects the health of mangrove forests and its ability to keep pace with rising sea levels. A clear example is the Fort-de-

France Bay where pollution is a severe problem (SAFFACHE 1999b; PUJOS ET AL. 2000).

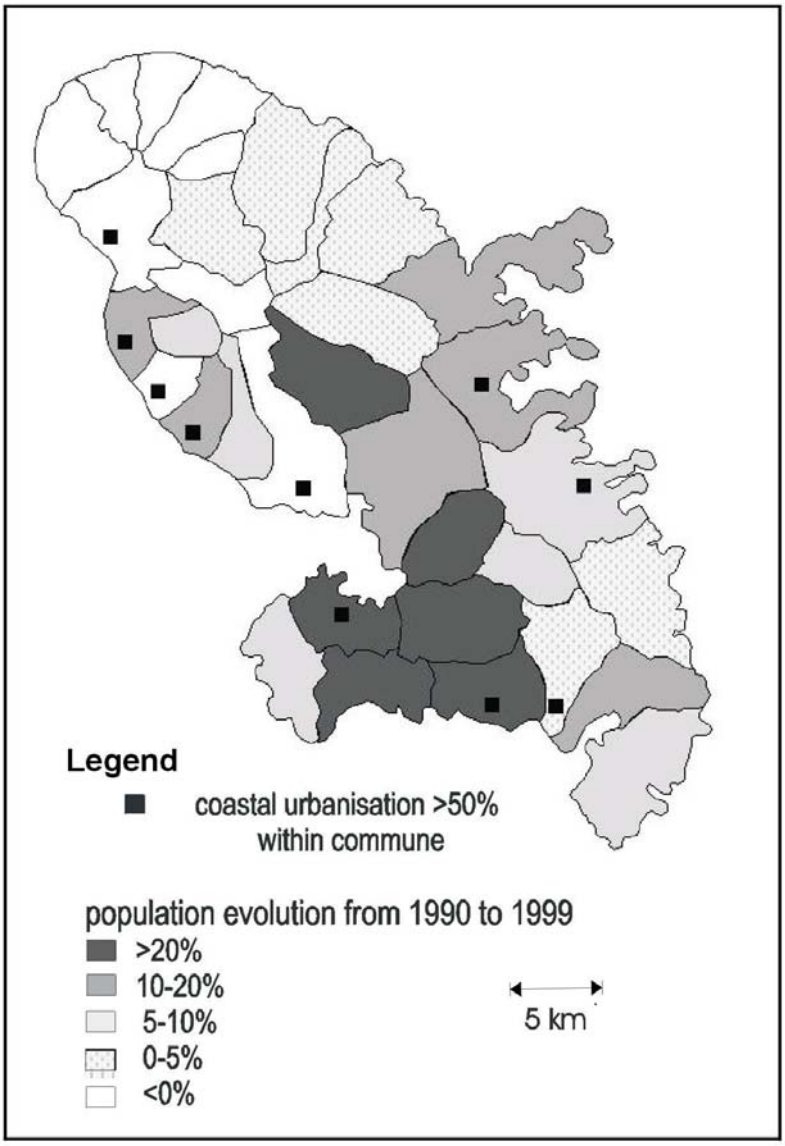


Fig 2.6 Evolution of district population and coastal urbanisation rates. Sources: GABRIE ET AL. 2004; ASSAUPAMAR 2002

2.3.3. Consequences of coastal squeeze and beach reduction for Martinique tourism

Local inhabitants depend on the diverse system of coastal and marine resources. The projected loss of beaches as a consequence of sea level rise may cause severe economic impacts on the tourism industry. A study on the beach-oriented island of Barbados showed that income from tourism is predicted to decline by 62% if the beach areas were significantly reduced (UYARRA ET AL. 2005). In

questionnaires tourists indicated a reduced probability of selecting the island destinations if climate change significantly altered features such as beach structure or coral reef health (UYARRA ET AL. 2005). The same may also happen on Martinique: about 85% of the beaches on Martinique are economically used for tourist recreation. That is a total length of nearly 42 km sandy beaches. Of these 27% are associated with a village, a sightseeing-road, or a harbour. The remaining majority are the lonely beaches that Martinique is famous for. Spatial analyses of potential tourist-attractive beaches revealed that only very few suitable beaches are not already used by the tourism industry. A total of 6 km of the beaches, especially along the northern coast, are out of reach, and the remaining beaches are not attractive for beach tourism. In total 83% of the tourist used beaches are at risk to coastal squeeze. If sea level rises beach reduction will become an increasing problem for the attractiveness of Martinique beaches as a tourist destination, because in general the Martinique tourist looks for the classic beach environment combined with warm climate, sandy beaches, an exotic picture, as well as relaxation opportunities. About 87% of the visitors of 2002 came to the island to relax on the beach (MARQUES 2003), for example. With rising sea levels the consequences might be similar to those of Barbados (UYARRA ET AL. 2005).

In addition to the consequences of beach reduction, infrastructure found up to 300 m landward of the beach and within an area below 5-10 m along the coast is vulnerable to accelerated sea level rise. That is an area of about 11-14% of the total island surface, where more than 62% of the infrastructure and about 53% of the total population is situated.

2.4. Summary

Spatial evaluation of the adaptive capacity of wetlands and beaches to sea level rise on Martinique shows that the regional wetland system is unlikely to collapse completely. Of the 79 km mangrove forests along the coast, only a total of 23 km is rated as highly sensitive to coastal squeeze, the majority of 55 km is of medium sensitivity, mainly due to high sedimentation rates. Furthermore, besides the migration opportunities and sediment budget the state of the ecosystem (degraded, stressed or in good condition) also needs to be taken into consideration. The marine and coastal ecosystems are stressed by agricultural pollution, hyper-

sedimentation of bays, urban and industrial pollution, especially due to sugar and rum production, as well as oil and petrol pollution (SAFFACHE 1999b). Studies about the coastal pollution of Martinique were available by SAFFACHE (1999b; et al. 1999; 2000). Degraded and stressed ecosystems may not be able to adapt to sea level rise impacts even if coastal squeeze is not a problem. Degradation makes the ecosystems more vulnerable to extreme events. Many Martinique wetlands, particularly the mangrove forests are overexploited and polluted. Figure 2.3 shows those districts of Martinique reporting coastal pollution and wetland degradation. Most of the districts are reporting severe coastal pollution. It is important to note that six districts at the south coast do not seem to have any pollution problems. These districts are the main beach tourist destinations. They depend on their untouched, natural image of white Caribbean beaches. Nevertheless, most of these districts report high mangrove degradation. Nearly all districts with mangrove-stands report degradation, except for Le Robert and La Trinité, where the nature park of "Caravelle Peninsula" protects the remaining mangroves from human destruction.

Losses of wetlands impact many sectors and functions of coastal areas including food production (loss of nursery areas for fisheries), flood and storm protection (storm surges will penetrate further inland), waste treatment and nutrient cycling, and the capacity to serve as a habitat for wildlife (NICHOLLS ET AL. 1999). The mangroves may be able to adapt to the changing conditions: *Rhizophora* mangle, the main mangrove species on Martinique, occurs in markedly different geographic habitats, under brackish, marine, and hypersaline conditions, as well as in developed estuarine fringe forest down to lower scrub (BACON 1993). A problem is the coastal and marine pollution that threatens the mangrove habitats and makes them more vulnerable to sea level rise impacts.

In addition to the probable mangrove losses, 45% of other coastal wetlands and more than 70% (40 km out of 57 km total) of the beaches are highly sensitive to coastal squeeze. In particular the fine sand beaches along the southern coast that serve as main tourist destinations are the most vulnerable to coastal erosion. Theoretically, shoreline migration will create new areas of economic benefit as new beaches are built, but because of the mountainous island character, steep shores, and anthropogenic constructions very close to the sea sedimentation

processes that might lead to beach evolution are unrealistic on Martinique. Therefore, the protection, replenishment and stabilisation of existing beaches, at least until major existing tourist investments are amortised, represents a principal socioeconomic goal (UNEP 1993). Against progressive coastal erosion it had become necessary for the regional council to work out defence strategies to protect the settlements and other infrastructure. Besides longitudinal and transversal buildings found along the coast at some settlements, breakwaters are built in front of hotel complexes in the south. Often the use of structural solutions interferes with sediment transport along the coastline and, with that, the shoreline stability of adjacent properties (UNFCCC 2000). Another aspect is that these measures are often not set in the right place to keep the view to the open sea for tourists. As a result protection for coastal erosion does not take place. In all, the protection buildings are not suitable for managing and protecting the coast permanently (UNEP 1989). Furthermore, no additional considerations and strategies for a potential rise of sea level and its consequences do yet exist for Martinique beaches and wetlands.

However, Martinique has more to offer to tourists than just sandy beaches: Since 1999 the island also promotes ecotourism as alternative to its «sea, sand, sun »-slogan. Ecotourism on the island includes nature and culture activities like mangrove excursions, bird watching, museums, a garden route and rum circle that shall make the inland island and mangrove forests more attractive for tourists. Between 1990 and 1998 the booking of rural accommodations by tourists has doubled. In 1999 already 3% of the Martinique visitors have been categorised as ecotourists. About 1 000 rural overnight accommodations have been counted in 1998 and future perspectives aim at 1 800 rural accommodations in 2006 and 2 800 accommodations in 2011 (NOSEL 2000).

Considering population developments on the island the northern districts might suffer from additional population loss, if coastal squeeze is going to continue further. The southern districts, however, are not going to decline in population, even if valuable beaches are lost. The main population growth factor is here the proximity to the urbanization zone of Fort-de-France. Moreover, population growth would increasingly happen in the hinterland that is agriculturally used to date. Taking this scenario together with growing ecotourism into account, land

use would change from agricultural fields for exports to suburban living spaces and rural tourist destinations. But the danger for the Martinique economy is still that the island may no longer be as attractive a destination as they is now and will lose visitors to competing destinations if environmental degradation, coastal squeeze, and beach reduction further continues as is projected here.

2.5. Conclusion

This study showed that spatial analysis is quite useful to locate and evaluate coastal parts sensitive to coastal squeeze and area reduction. On the basis of several spatial datasets, Martinique beaches, mangrove forests, deltaic and estuarine areas, as well as coastal swamps, are examined with regard to the risk of coastal squeeze and area reduction. The spatial evaluation of the adaptive capacity of wetlands and beaches to sea level rise on Martinique revealed that the majority of the beaches are highly vulnerable to area reduction, whereas only 29% of the mangrove forests are rated as highly sensitive to coastal squeeze. The majority is of medium sensitivity, mainly due to high sedimentation rates. But many Martinique wetlands are also overexploited and polluted. The state of the ecosystem is a factor that affects coastal squeeze and should be included into the sensitivity modelling as soon as appropriate data are available not to underestimate the results.

This study also tried to address the correlation between human impacts and wetland reductions. Often not only topography prevents the wetlands to shift inland, but also anthropogenic infrastructure. Especially the tourism industry often occupies areas very close to the sea. On the other hand the same is also dependent on the existence of wide natural beaches. The key aspects of the popularity of Martinique as a tourist destination are its fine sandy beaches, clear water and pristine habitats. If accelerated sea level rise further continues, Martinique is endangered to loose not only the majority of its famous beaches and its valuable mangrove habitats but also its prestige as beach tourist destination.

The data situation on Martinique is quite poor regarding coastal squeeze. It is therefore important to undertake more detailed studies to improve this littoral vulnerability assessment. Nevertheless, this study is a suitable attempt to

emphasize that not only are low lying islands exposed to the consequences of accelerated sea level rise, but that also mountainous islands are vulnerable.

3. Spatial assessment of sea level rise on Martinique's coastal zone and analysis of planning frameworks for adaptation

3.1. Introduction

3.1.1. Climate Change and Sea Level Rise in the Caribbean

During the last century, a relative sea level rise of about 20 cm has been observed in the Caribbean (MAUL 1993), and its speed is increasing. Relative sea level was estimated to rise on average 2.8 to 5.0 mm/year during the 1990s (MAUL 1993). Therefore, regional projections state a rise in sea level of 10 to 50 cm by 2025 as realistic (MAUL 1993; IPCC 2001). Additionally, Climate Change scenarios project an increasing frequency and intensity of hurricanes and tropical storms for the Caribbean region (UNEP 2000; IPCC 2007), both causing coastal flooding and higher erosion rates at the shores. Accelerated sea level rise will have enormous consequences for the coastal structures through flooding, inundation, erosion, recession of barrier beaches and shorelines destruction and drowning of coral reefs and atolls disappearance or redistribution of wetlands and lowlands, increased salinity of rivers, bays, and aquifers, and loss of beaches and low islands. An extension of the coastal hazard area is also expected due to the combination of accelerated sea level rise with natural disasters (UNEP 2000; IPCC 2007). Besides the loss of natural coastal structures, man-made measures might get affected with greater populations at risk in low lying areas as could have already been observed in the region during the last few years.

3.1.2. Martinique and its coastal population

The economy of the Lesser Antilles' island Martinique is largely based on the export of agricultural goods (bananas, sugarcane, and pineapples) and tourism as major income sources. Nearly one million visitors arrive each year on the island that is inhabited by nearly 400 000 people (MARQUES 2002; CHARRIER 2003). Because of its mountainous terrain, the majority of the settlements and about 77% of the population are situated along the coast below 20 metres. Neglecting security, most of the houses were constructed very close to the shoreline. The urbanisation of Martinique was characterised by a flux from the inland to the

littoral and the concentration of population in one extending urbanisation zone. Fort-de-France is the biggest agglomeration area of the island and the pole of development. Here more than 43% of the total population live in 15% of the island's surface area (GÉNIX AND LAMPIN 2003) almost at the level of the sea. Today, migration fluxes from the inland island to the littoral are still observed (HOCREITÈRE 1999; WILLIAM 2000). But due to a rising standard of living as well as better infrastructure and mobilisation by car, a suburbanisation to the inland and to the southern districts also takes place. Riviére Salée, for example, showed a growth of more than 40% (DELBOND ET AL. 2003). The northern island on the contrary is characterised by demographic and economic decline. The populations of the four communities in the extreme north (Grand Riviére, Prêcheur, Sainte-Pierre, Macouba) shrank the most: 10.34% between 1990 and 1999 (DELBOND ET AL. 2003; SEE ALSO GÉNIX AND LAMPIN 2003). This region suffers from insufficient infrastructure and rough terrain. The main economic activities here are export agriculture and fisheries (WILLIAM 2000). The growing population of Martinique - in 2003 the annual population growth rate amounted to 1.4‰ (IFRECOR 2003) – additionally extents the coastal urbanisation.

3.1.3. Policy instruments for the coastal zone on Martinique

It is important for adaptation strategies for the coastal zone to consider the essentials of the local coastal zone management plans and the corresponding policy instruments. These consist of regional and national but also EU-wide regulations because the Caribbean Lesser Antilles' island Martinique is a French Department (DOM - Departement d'Outre Mer) and therefore an EU „ultra-peripheral region". This section gives an overview of the most important legislation instruments for the coastal zone of Martinique.

« *La loi des 50 pas géometriques » and its colonization.* On Martinique the littoral is characterized by a zone called "les 50 pas du Roi" or "cinquante pas géometrique", that means a zone of 81.2 m from mean high water tide level landwards (HOUDART 2004). After the "loi littoral" this stripe is today part of the public domain of the state. On Martinique the "50 pas" represent 3 513 ha of which 35% are under intensive human use (public institutions, tourism, agriculture, fisheries, artisans, industries). The cause of the high population

density within the 50 pas lies in Martinique's coastal zone management history: From 1922 until 1955, the privatisation of the 50 pas was enforced. From 1955 onwards the zone was again integrated into the public domain of the state. However, parcels of coastal land still have been sold – only half-legal - and until today the littoral is still seen as privileged space for houses. Additionally, the illegal occupation of the littoral without landholding for the economic reasons has been practised, when the sugar crisis and following concentration in (urban) tertiary activities took place. The development of agglomerations and diffuse habitats along the coast caused many problems. Therefore, plans have been formulated to regulate and limit the urbanisation, the tourism and industry for a protection of the remaining natural zones. In 1962, 65% of the coastal zone has been placed under the control of the ONF (Office national des forêts) and finally in 1986 the "loi littoral" merged the 50 pas into the "public domain maritime". That entails that urban areas within this zone are reserved for necessary public installations, for economic activities, or for general utilisations of the sea. Urbanised areas within the 50 pas cannot be build on if they are used as beach, forest, garden, or park.

The "loi littoral" on Martinique. The most important law concerning the coastal zone on Martinique is the so called "loi littoral" (FRANCE GOUV. 1986). It was elaborated in 1986 by the «Direction du transport maritime, des ports et du littoral», and by the « Direction générale de l'urbanisme et de l'habitat et de la construction », under collaboration of numerous French ministerial departements. It has been transmitted to Parliament in 1999. The regional objectives for the coastal zone described in the "loi littoral" are (ALDUY AND GÉLARD 2004):

- research and innovation of particularities and resources;
- protection of biological and ecological equilibrium, erosion mitigation, preservation of sites and landscapes;
- extension of urbanisation only within those sectors that are today occupied by diffuse urbanisation;
- prohibition of constructions and utilization of slopes adjacent to the littoral, if they blur the character of the landscape;
- preservation and development of economic activities in relation to the sea, like fisheries, aquaculture, ports activities, ship construction and reparation

and marine transport; for example, construction of new ports of pleasure is curbed, therefore existent ports shall be extended; and

- maintenance and development of agricultural activities or forestry, of industries, crafts, or tourism within the coastal zone.

SMVM (schémas de mise en valeur de la mer) and SAR (schémas d'amenagement régionaux). Regional Management schemes (SAR) additionally regulate the utilization of the coastal zone for tourism, constructions and commercial use. In France, the state is traditionally responsible for coastal protection, but since the law of decentralisation (1984) the decisions for coastal management are in the hands of the regional councils («departements»). Its implementation is presented in the «Schéma de Mise en Valeur de la Mer (SMVM) ». The SMVM gives a high priority to protective measures: protection policies for the coastal strip concern natural coastal areas, areas of outstanding interest designated for protection (Etang des Salines, Morne Jaqueline, Caravelle, and the Lamentin mangrove swamp) and urban development buffer zones. In the DOM-TOM the SMVM are replaced by regional management schemes, the SAR. The SAR (Schémas d'Amenagement Régionaux) are elaborated and adopted by the Départements d'Outre-Mer and have to be accepted by the National assembly. Martinique has had SAR since 1998 (HOCREITÈRE 1999). Planning policies on Martinique focus mainly on the regulation of urbanisation and town planning as well as on provisions to improvements of urban wastewater and rainwater run-off treatments. The SAR are jurisdictionally situated between the "loi littoral" and other regulations of urbanisation (Schémas de coherence Territoriale, plans Locaux d'ùrbanisme). They are seen as an orientation document and tool for integrated coastal management, for administration and sustainable development of activities.

As a French department, Martinique is a European territory in which most European Union agreements, directives and laws are applicable, as well as those rules that are more specifically designed for outlying EU regions such as the DOM-TOMs (cf. EUROPEAN COMMISSION 2007).

3.1.4. Evaluating vulnerability and adaptation to sea level rise

"Vulnerability is the extent to which a natural or social system is susceptible to sustaining damage from Climate Change" (IPCC 2001). A study by the World

Bank (DEEB 2002) criticises the lack of adequate data to conduct vulnerability assessments in the Caribbean. There is a need for vulnerability studies along those coasts where the data availability is bad (KLEIN ET AL. 1999; KLEIN & NICHOLLS 1999; DEEB 2002). Also the Common Methodology for Assessing Vulnerability to Sea Level Rise, which was developed by the former Coastal Zone Management Subgroup of the IPCC (IPCC CZMS 1992), requires accurate and complete data. This has limited its applicability and made the development of alternative assessment methodologies necessary, and has led to a lack of vulnerability studies where accurate quantitative data are missing (KLEIN AND NICHOLLS 1999). However, accelerated sea level rise already affects the Caribbean coasts and there is a need to formulate risk and vulnerability assessment methodologies compatible with the data available. The IPCC (2001; PARRY ET AL. 2007) even declares that one of the most important climate change effects on coastal resources will be sea level rise. Small islands will be especially vulnerable to the effects of Climate Change (MIMURA ET AL. 2007). VOLONTE AND NICHOLLS (1999) give a first overview of how to conduct vulnerability assessments in the Caribbean region. LEWSEY ET AL. (2004) call for increasing use of GIS and remote sensing to obtain useful results. THUMERER ET AL. (2000) conducted such a successful GIS assessment for the English east coast, for example. For this study, a GIS-based assessment model has been developed, that allows spatial explicit assessments of coastal vulnerabilities. The methodology should ensure easy application to other coastal zones by utilisation of parameters that can be derived through GIS and always considering the individual characteristics of different coastal areas. Most sea level rise impact studies concentrate on low lying shallow costal zones only (e.g. KONT ET AL. 2003; GAMBOLATI ET AL. 2002) and neglect the impact potential of hurricanes and sea level rise to coasts with mountainous topography. Therefore this study intentionally chose the mountainous Caribbean island Martinique as case study site.

The coastal zone of Martinique is a very diverse space, partly occupied by human constructions, including tourist resorts, and partly by valuable ecosystems. It is surprising that, on Martinique, present sea level rise is not addressed in coastal management even though saltwater intrusion and coastal erosion is locally already a severe problem. On Martinique, where most of the settlements are situated along

the coast and beach tourism is the main source of income, a change in coastline and an extension or intensification of the impact area might have enormous effects on the island's economy, not to forget ecological consequences such as wetland loss, etc. The concept of coastal vulnerability to erosion and inundation encompasses more than the exposure and sensitivity of natural and human systems to potential impacts of climate change. It is also defined by the degree to which these systems can prepare for and respond to impacts (KLEIN 2002). Coastal adaptation is assessed in different ways and at different scales as varieties of studies show. Some studies are conducted as integrated model framework (e.g. DINAS-COAST CONSORTIUM 2006), others concentrate on specific coasts (SNOUSSI ET AL. 2008), ecosystems (GILMAN ET AL. 2008), or on the economic effects of sea level rise (DARWIN AND TOL 2001; NICHOLLS AND TOL 2006; TOL 2007). Despite recent research progress made in this field the IPCC Report (NICHOLLS ET AL. 2007) still mentions research gaps in the development of methods for identification and prioritisation of coastal adaptation options.

Therefore it was not only of importance to model the spatial impacts of sea level rise but also to evaluate its possible consequences and discuss potential and existing mitigation and adaptation strategies for Martinique. Initially, there was a need to describe the actual situation and legislation measures for coastal zone management of the island. The determination of potential adaptation strategies should provide a valid tool for coastal zone planning and management. A specific methodology for this regional-scale vulnerability assessment has been developed in this study. The methodology should only rely on spatial data that can be evaluated from satellite data or topographical maps. This has the advantage that it is better adapted to local needs. However, the GIS-based approach ensures easy applicability to other coastal zones by allowing regional variations to be considered.

3.2. Methodology to conduct spatial planning assessments

The methodology is divided into three parts. The first part evaluates the vulnerability of the coastal resources to sea level rise, the second investigates existing and potential coastal zone management strategies for formulation of policy targets, and the third part describes the spatial translation of suitable

adaptation strategies via GIS. Figure 3.1 gives an overview of the applied methodological structure.

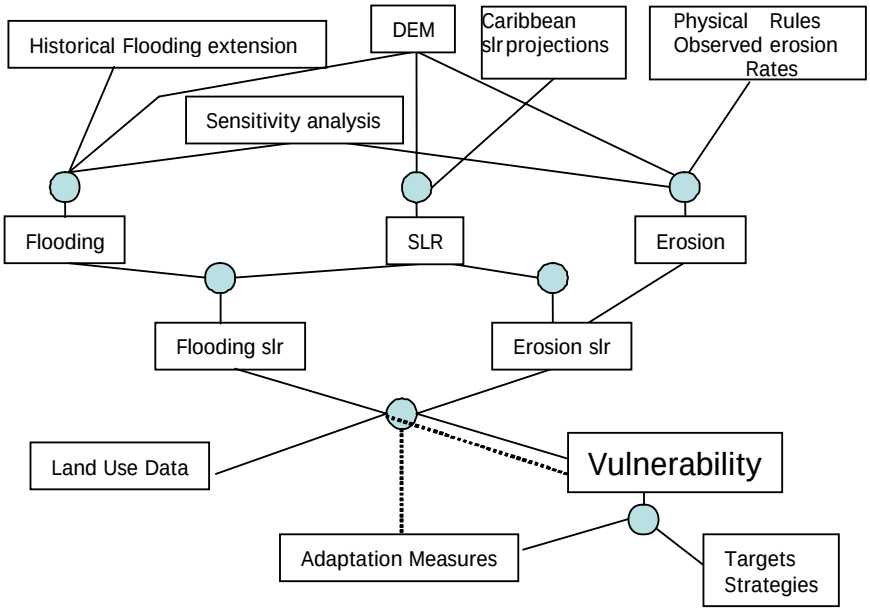


Fig 3.1 Structural overview of the Methodology.

3.2.1. Vulnerability evaluation to sea level rise impacts

The aim of the first part of this study is to illustrate the consequences of accelerated sea level rise for the inhabitants of Martinique. Therefore, a GIS-based model has been developed in Model Builder of ArcGIS 9 that delineates the potential coastal hazard areas with help of sea level rise scenarios.

The main threats to the coastal zone are flooding and erosion. Shallow land in the Caribbean is especially sensitive to flooding and erosion during hurricanes or tropical storms. SCHLEUPNER (2007) evaluates the present coastal hazard areas on Martinique to erosion and inundation during hurricanes through a spatial model. This model has now been used as the base for the sea level rise impact study. If the sea level rises, the flooding risk will shift to higher elevations and would additionally cause erosion and inundation (NICHOLLS ET AL. 1999; UNEP 2000).

Two sea level rise scenarios have been chosen out of the IPCC scenarios and regional sea level rise projections (MAUL 1993; IPCC 2001) and applied for Martinique. These scenarios state a rise in sea level to 2100 of 50 or 100 cm. The sea level rise scenarios are added to the flooding and erosion scenarios of the GIS

model. The SRTM3 (Version 2) digital elevation model of Martinique interpolated with digital topographical data of the coastal zone (IGN 1996) was used.

The erosion and flooding scenario model additionally uses the following rules and remarks: According to BEHNEN (2000), areas below 10 m level are most vulnerable to sea level rise. Hereby, more shallow slopes experience a greater increase in flood risk due to sea level rise than steeper slopes (NICHOLLS ET AL. 1999). BRUUN (1962) showed that, as the sea level rises, the upper part of the beach is eroded and the material is deposited offshore in a fashion that restores the shape of the beach profile with respect to sea level. The "Bruun Rule" implies that a rise of one meter would generally cause shores to erode 50 to 200 meters along sandy beaches. Coastal wetlands or muddy coasts would become even more vulnerable to erosion: unlike sand, muddy sediments can be carried great distances before dropping out of suspension. On this basis, UNEP (1989) projects a shoreline retreat for each centimetre of sea level rise up to several meters horizontally. Data on observed erosion rates and historical flooding extensions[1] are also used as "experience" values of the model and serve for validation purposes. As a result we obtain a spatial assessment of the sensitivity of the coastal zone to sea level rise, flooding and erosion risk as well as its impact area.

The results of the flooding impact area evaluation through Coastal Sensitivity analysis are translated into five graded rating classes from extremely high sensitivity to no sensitivity expressed through the F- index as explained in table 1. Whereas the F-index gives information about the impacted area through flooding at sea level rise, the erosion risk is also of importance. Therefore, an index value for the erosion risk has been added (E). Not only the low lying coastal parts might be affected by sea level rise, but also higher areas at risk of enhanced cliff erosion. Through the consideration of flooding and erosion risk both effects can be taken into account separately or combined.

Land cover and socio-economic geo-data are included into the model to get information about human vulnerability. These are obtained from interpretation of

[1] ASSAUPAMAR 2002; METEO-FRANCE 2000; PUJOS ET AL. 2000; SAFFACHE 1998; SAFFACHE ET AL. 1999; SAFACHE AND DESSE 1999; SAFFACHE 2000; SAFFACHE ET AL. 2002

satellite images (LandSat), topographical maps (IGN 1996)[2], and statistical data[3]. We used the distribution of beach hotels, tourist destinations including beaches, human settlements, houses and population densities, as well as harbours, coastal industries and other infrastructures as parameters that were intersected separately into the impact area. First of all, maps of the population densities within the impact area and infrastructure data have been created by intersection. To obtain statements about the vulnerability of the population and infrastructure, the data were translated into a 5-level assessment scheme. Table 3.1 shows the description of the parameters and their scaling for the example of population density and infrastructure (D-index).

Table 3.1 Scaling and description of the index parameters.

Index	Flooding Risk (F)	Population/Infrastructure Density (D)	Erosion Risk (E)
1	Very high flooding risk (flooding at every storm event under present conditions)	Very high densely settled areas (>750 Inhabitants/km²), also harbours, ports, industries	Very high erosion risk (under present and future conditions even without storm event)
2	High (flooding at storm events from category 2 onwards under present conditions or at any storm event under slr scenarios)	High densely settled area (250 – 749 I/km²), important infrastructure	High (under present and future conditions at any storm event)
3	Medium (flooding at storm events from category 3 onwards under present conditions or from category 1 or two storms under slr scenarios, no flooding during tropical storms)	Medium settlement (100-249 I/km²) and infrastructure density	Medium (erosion only under slr scenarios and any storm events)
4	Low (flooding only at extreme events like tsunamis, or under hurricanes with intensities of 4 or 5 at all scenarios)	Sparely settled (20-99 I/km²), agricultural use, few infrastructure	Low (rock resistance against erosion high, erosion only under slr scenarios and during extreme storm events)
5	No flooding risk (at any scenario)	Negligible human utilization (0-19 I/km²)	No erosion risk (at any scenario)

[2] Landsat Data used from www.geocomm.com
SRTM3 (Version2) Data from EastViewCartographic: www.cartographic.com
Other spatial data: www.geoportail.fr

[3] Statistical data are obtained from www.martinique.pref.gouv.fr, as well as from CHARRIER 2003; CONSEIL REGIONAL; INSEE; MARQUES 2002, STATISTIQUE-PUBLIQUE

The three parameters build the basis for the vulnerability evaluation that is expressed through the five-levelled vulnerability index (VB) with “1” meaning highest vulnerability of erosion and inundation considering sea level rise. The assessment relies on logical constraints that are shown in the following equations:

$$VB1 = (F_1 \vee E_1) \cap (D_1 \vee D_2)$$
$$VB2 = (F_2 \vee E_2) \cap (D_1 \vee D_2)$$
$$VB3 = [(F_3 \vee E_3) \cap (D_1 \vee D_2)] or [(F_1 \vee F_2) \cap D_3] or [(E_1 \vee E_2) \cap D_3]$$
$$VB4 = [(F_4 \vee E_4) \cap (D_1 \vee D_2 \vee D_3 \vee D_4)] or [(F_1 \vee F_2 \vee F_1) \cap D_4] or [(E_1 \vee E_2 \vee E_3) \cap D_4]$$
$$VB5 = (F_5 \vee E_5) or [(F_1 \vee F_2 \vee F_3 \vee F_4) \cap D_5] or [(E_1 \vee E_2 \vee E_3 \vee E_4) \cap D_5]$$

$\vee -$ logical"or"
$\cap -$ intersection

There are numerous studies that transform complex data sets into indices in order to assess the sensitivity of areas to threats (COOPER AND MC LAUGHLIN 1998; KLEIN AND NICHOLLS 1999), and to define coastal vulnerabilities to sea level rise. This has either been executed as a function of coastal erosion, or by variation of sea level or in an ecological and cultural context (GORNITZ 1991; LIU 1997; KLEIN ET AL. 1998; FRIHY ET AL. 2004). Coastal vulnerability indices are often used as management tools at different spatial scales (KONT ET AL. 2003; VAFEIDIS ET AL. 2004; SNOUSSI ET AL. 2007).

The logical assignments have been chosen on basis of Boolean Logic and Map Algebra. In nature conservation, this is widely practiced (BLASCHKE 1997; LANG AND BLASCHKE 2007; LINDENMAYER AND HOBBS 2007) and is preferred over arithmetic assignments of single assessments through average determination, for example. The latter as well as the use of additional weighting factors often simulate pseudo-objectivity only. Logical assignments prevent this.

The level “VB1” consists in this case of those areas that show very high erosion- or flood risk and very high to high settlement density (for scaling see table 1). On the other hand is the vulnerability level “5” characterized by negligible erosion or flooding risk or alternatively, by any erosion or flooding risk and no human utilization. VB3 is reached either through medium erosion- or flooding risk and very high to medium settlement density or through very high to high erosion- or

flooding risk and medium settlement density. As a result, vulnerability maps for each human coastal resource illustrate the corresponding vulnerability to the effects of sea level rise. The results also allow further analysis in combination with adaptation strategy evaluations.

At the moment, all artificial measures of the coast are excluded from the model. A methodology is described below to apply these measures and potential additional adaptation measures to sea level rise to obtain more realistic statements about the vulnerability to sea level rise impacts.

3.2.2. Formulation of Coastal Zone Management strategies and targets

After evaluating the vulnerability of the human coastal resources there is a need to define targets for coastal zone management practices concerning sea level rise effects. The objective of this part of the methodology is to discuss coastal zone management strategies by describing the actions and measures undertaken concerning accelerated sea level rise on Martinique. The investigations of Climate Change/Sea Level Rise response strategies are based on intensive literature review (CGCED; CAMBERS 1992; BRAY ET AL. 1997; NURSE 1997; CPACC 1999A&B; VOLONTE AND NICHOLLS 1999; CPACC 2000; PHILLIPS AND JONES 2006). The evaluation of the coastal zone management strategies in combination with intensive literature review and the results of the sensitivity and vulnerability assessments described above form the base for formulation of policy options and targets for the entire coastal zone of Martinique. Any of these targets might be realized by several defined adaptation strategies.

3.2.3. Development of Adaptation Potentials

In the last step, the most suitable adaptation measures per coastal segment are evaluated through the targets and the evaluation of vulnerability. For translation of the targets into a GIS we assume that the adaptation strategy also determines the adaptation measure. Depending on its vulnerability, geomorphology and land cover the formulated targets can be determined for each coastal segment. That means, for example, that only those coastal parts are considered for protection strategies that demand those measures by high vulnerability. The evaluation of adaptation strategies is carried out for each vulnerability parameter separately. As a result, adaptation maps are obtained concerning the vulnerability of different

coastal resources. Dynamic interaction occurs in that way that the natural system impacts on the socio-economic system and planned adaptation by the socio-economic system influences the natural system (NICHOLLS 2003). Concluding, adaptation might reduce the impacts of sea level rise and climate change (BURTON ET AL. 1998). KLEIN ET AL. (2001) give an overview of the technological options for adaptation to climate change in coastal zones and the latest IPCC Report (PARRY ET AL. 2007) addresses adaptation to climate change impacts as an important issue in future coastal development.

3.3. Results

3.3.1. Sensitivity and Vulnerability evaluation to sea level rise impacts

The evaluation revealed that the coastal sensitivity to flooding and erosion increased with rising sea level in comparison to present conditions whereas the spatial distribution of sensitive coastal segments generally remained the same. SCHLEUPNER (2007) shows that, under present conditions, 13% of total coastline of 432 km is rated with low sensitivity, 43% have medium sensitivity, and 44% show a high risk of coastal flooding and erosion. Knowledge of the hazard area is important for the evaluation of vulnerability. The extension of the impact area serves as base for the vulnerability evaluation of human resources. The coast is especially attractive for residential, economic and for tourist activities. The spatial analysis showed that tourism infrastructure, road networks and major settlements are usually all located along the coast, giving locals and visitors an easy access to the coastal and marine natural resources. Analyses of the present impact state to flooding show that 58 km² have a very high flood risk, 55 km² lie in the range of high impact risk, and 57 km² reveal medium risk. Altogether, this amounts an area of 170 km² or about 16% of the islands surface. More than 62% of the infrastructure and half of the Martinique population (53%) are situated within this zone. The spatial evaluation of the impact extent of accelerated sea level rise identifies the areas that are likely to be affected by flooding and erosion depending on the scenarios of sea level rise. In total, 106 km of coastline would be affected by erosion if sea level continues to rise up to 50 cm, mainly along the north-western island's coast. This is about one fourth of the coast including an assumed 500 m landward impact zone (CAMBERS 1997). Additionally the flood

impact area has been determined. A sea level rise of 50 cm enlarges the flood impact area to 221 km² or 20.5% of the total area: 68% of the infrastructure and 65% of the total population would be affected. This is a total population number of about 260 000. More than 36% of the impact zone is attributed with the category “expansion area” of settlements. An evaluation of infrastructure and constructions situated within this zone reveals that settlements along the southern coast are seldom found below an elevation of 5 m whereas at the northern coast they reach further down to sea level. But also tourist hotels can be found very close to the sea and below the 5 m level. However, the majority of coastal constructions are built on average at heights between 5 and 10 m above the present sea level and therefore within the impact zone of flooding and erosion.

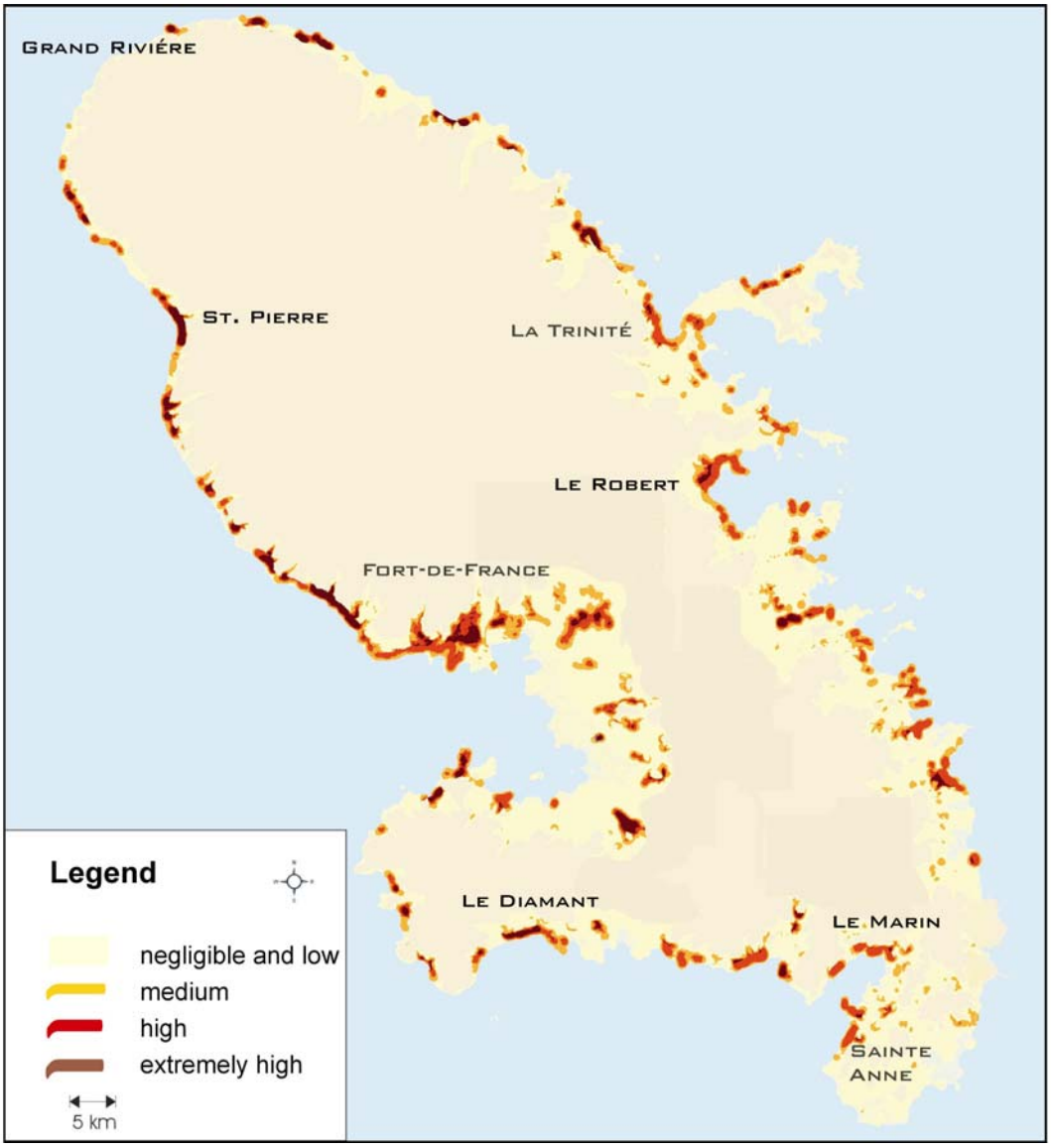


Fig 3.2 Vulnerability to sea level rise and its impacts concerning population density.

Figure 3.2 illustrates the vulnerability of affected human coastal population. The greatest expansion of the coastal impact areas can be found in the Fort-de-France Bay and at the bays of the south-western island. These areas are also those parts of the island where high population numbers and settlements are concentrated. People and their houses are therefore very vulnerable. But also in the northern half of the island, the human developments are situated right between the sea and the steep slopes of Mt. Pelée. These small, narrow areas show highest sensitivity to flooding and erosion at any scenario. Here, the small land adjacent to the beach is often the most densely settled area, because it is the only flat land available. Table 3.2 illustrates that the space of settlement is limited by topography. The distribution of population in relation to its relative height to present sea level shows that elevations below 100 m are densely settled and that the majority of population lives at slopes of less than 10%.

Table 3.2 Distribution of population in relation to its relative height to present sea level.

altitude (m)	**area (km^2)**	**Pop. (inh)**	**Popdens (inh/km^2)**	**area (km^2) slope <10%**	**Pop. (inh.) slope < 10%**	**Pop. dens slope < 10%**
0-10	117	79 500	679	96	78 000	812
10-100	363	255 500	704	288	221 000	767
> 100	628	64 000	102	277	61 500	222
total	1 108	399 000	360	661	360 500	545

Martinique as a tourist destination is famous for its fine sandy beaches, clear water and pristine habitats. 13% of the total coastal area consists of sandy beaches. But the fine sands beaches along the southern coast that serve as main tourist destinations are the most vulnerable to coastal erosion during hurricanes. On Martinique, 62% of all beaches and 66% of tourist used beaches are at risk of erosion. That means also that the erosion rate is higher than the rate of accretion. In addition, especially the tourism industry often occupies areas very close to the sea, often even below 5 m in former mangrove areas. These constructions have a very high flood risk during hurricanes. The lost mangroves forests used to serve in erosion and flood protection. Altogether, 80% of the coastal hotels and tourist

resorts including camping areas are at risk as well as 92% of the main coastal tourist destinations without overnight stay possibilities like small islets, fishery settlements, lonely beaches, for example.

Distilleries and sugar refineries are the main businesses besides tourism. Only a few are found in the impact area of inundation, as the great majority is situated in the hinterland. Nevertheless, the coast of Martinique is attractive for industrial developments as well. Especially the Fort-de-France bay with its extensive docks providing space for diverse industries (e.g. chemicals, construction). An analysis of the locations of vulnerable industries of Martinique reveals that places at St. Pierre, Fort-de-France and at La Trinité are the most at risk to flooding during hurricanes.

3.3.2. Coastal zone management on Martinique - legislation and response strategies to accelerated sea level rise

After the evaluation of human vulnerability, we turn to Martinique's coastal zone management strategies and its adaptation plans to accelerated sea level rise, intensified erosion and inundation. This leads to the formulation of goals for future coastal zone management concerning sea level rise.

In France all levels of government have their role in developing planned adaptation measures. The coastal zone management of the Départements d'Outre Mer (DOM) mentions several coastal response strategies. These are the protection measures ("défense rigide"), but also accommodation and planned retreat strategies ("défense souple") (see also DENEUX 2002). The following explanation of coastal zone management strategies on Martinique refers to the definition of the terms by KLEIN (2002). In practice, many response strategies are hybrid and combine approaches (NICHOLLS 2003).

Accommodation or planned retreat ("défense souple"). With (planned) retreat, all natural system effects are allowed to occur and human impacts are minimised by pulling back from the coast. With accommodation, human impacts are minimised by adjusting human use of the coastal zone (NICHOLLS 2003). The accommodation or planned retreat concept accepts and integrates natural coastline evolution into conservation plans. Also accelerated sea level rise is tolerated here. On Martinique a Water Management Masterplan (SDAGE) has been completed in

1999. Here, coral reefs, seagrass beds and mangroves are taken into account as sensitive areas. Especially the Conservatoire du littoral favours the défense souple along parts of the Martinique coast, where protection measures shall be avoided. The Conservatoire du littoral (Conservatoire de l'espace littoral et des ravages lacustres) is a public organisation with the remit of ensuring the definitive protection of outstanding natural areas on the coast, banks of lakes and stretches of water of 1 000 ha or more (BOYER 2000). These are mainly the natural and especially the protected parts of the Martinique coastline and less the highly populated areas. Martinique has several protected land areas (Regional Nature Park, Caravelle Peninsular and Sainte Anne islets, the Montagne Pelée, the Rocher du Diamant) bordering the sea. The Regional Nature Park comprises two separate areas that constitute 60% of the island's surface of Martinique. It includes the mountainous, volcanic part of the island, but also coastal cliffs, lagoons, and beaches. It excludes the cultivated lowlands. Other areas with nature protection include the Rocher du Diamant and Cap Salomon. The Coastal and Lakeshore Conservation Agency (CELRL) has purchased six areas totalling 1 135 ha on Martinqiue (Pointe Rouge/Trinite, Caravelle/Trinite, Grand Macabou/Marin-Vauclin, Morne Larcher/Anses d'Arlet-Diamant, Cap Salomon/Anses d'Arlet and Anse Couleuvre/Precheur). However, not only nature protection sites but also other utilized areas might be managed through the accommodation concept. The Conservatoire states that it should not be necessary to intervene at present into utilized zones that might only be impacted in 50 years time. The French Senat on the other hand sees a need to assess at least the future potentials of these coastal zones according to their potential for future cultivation.

Protection ("défense rigide"). Protection means that natural system effects are controlled by soft or hard engineering, reducing human impacts in the zone that would be impacted without protection (BIJLSMA ET AL 1996; KLEIN ET AL 2001). Such protection measures are the main response strategies against erosion and inundation in France (DENEUX 2002). The legislation of France manages a total coastline of 6 959 km (5 500 km continental and 1 459 km outre-mer). About 35% (1 925 km) of the French coast consists of beaches, and 21% of these beaches are artificially protected by measures (DENEUX 2002). The central government gives subsidies for coastal protection. In addition, it coordinates the politics about

“protection and prevention of the coast” (“PPR littoraux”) of the districts. On Martinique it has also become necessary for the regional council to develop defence strategies against erosion to protect the coast. But the operations to protect the inhabited places from the sea are complex and a single technical solution does not exist. Three types of buildings are common on Martinique: longitudinal (made of cement and concrete) and transversal (made of basalt rocks) constructions, as well as breakwaters. The communities of Lorrain, Marigot, Precheur, Diamant, and St. Anne use the first type, whereas the transversal buildings can only be found at Tartane. Breakwaters are mostly built in front of hotel complexes in the South. Moles, piers and other docks that absorb wave energy are also considered as protection measures. These measures might be effective, but they are expensive. Besides this, the changed wave actions and currents have negative influences on the environment. Naturally, the beach receives sediments from rivers and from the sea to compensate for the losses incurred by waves. The use of structural solutions interferes with the sediment transport along the coastline and, consequently, the shoreline stability of adjacent properties (UNFCCC 2000). To manage and protect the coast permanently the protection buildings are therefore not suitable (UNEP 1989). An alternative or supplementation to the protection buildings is beach nourishment („artificiel rechargement“) at suitable locations (PHILLIPS AND JONES 2006). The revenue generated from beach tourism might be used to finance this expensive measure. However, environmental impacts have not been properly studied yet (GREENE 2002).

Education, training, Public Awareness. In addition to the above mentioned the information strategy is of great importance. Public awareness and the development of evacuation plans should be included in every adaptation strategy. In the Caribbean many island states formed alliances and partnerships to elaborate coastal zone management or hazard evacuation plans, and to formulate climate change mitigation strategies (for example, CPACC, OGCED). However, the French islands of Martinique and Guadeloupe are relatively isolated in the Caribbean. On Martinique, formulation of targets concerning sea level rise and even the evaluation of the impact areas are missing as well as adequate public information.

After intensive study of the Martinique coastal zone management legislation (see above) and comparisons with other studies (CAMBERS 1992; BRAY ET AL. 1997; CPACC 1999A&B, CPACC 2000; CGCED 2002; KLEIN 2002; LEWSEY ET AL. 2004) policy options for the coastal zone of the island concerning accelerated sea level rise can be formulated. The targets for Martinique are:

- protection of existing or rehabilitation of degraded mangrove forests that have the capacity to reduce the impacts of natural hazards;
- accommodation to rising sea levels of natural areas;
- creation and maintenance of buffer zones / set back areas between land and sea where safety is not guaranteed;
- relocation or abandonment of settlement/infrastructure only if existing safety standard is not maintained, people directly affected agree, and the coastal defence administration is kept free of extra costs;
- prohibition of new buildings, modern estates or hotels in the highest impact areas;
- conditional reconstruction: existing houses in high impact areas shall not be rebuild if destroyed;
- only industrial or commercial use permitted within highest impact areas;
- protection of densely settled coastlines with hard and soft structures;
- strengthen of risk awareness of coastal population;
- development of public evacuation plans considering sea level rise;
- protection of economically valuable beaches from erosion only by measures of low habitat impact.

3.3.3. Illustration of Adaptation Potentials

The targets as well as the results of the vulnerability analysis serve as base for the development of a GIS-based model that is able to illustrate the potential distribution of adaptation measures. One map has been created for each vulnerability factor. Figure 3.3 shows the adaptation measures concerning the vulnerability of the population with respect to its density. About 18% of the total coastline therefore needs to be protected by hard measures, whereas another 15% or about 78 km of the vulnerable coast could adapt to rising sea levels by mangrove forest conservation and regeneration. The remaining coastline might

serve well with accommodation even if along those 93 km scattered houses or small settlements are found within the impacted area.

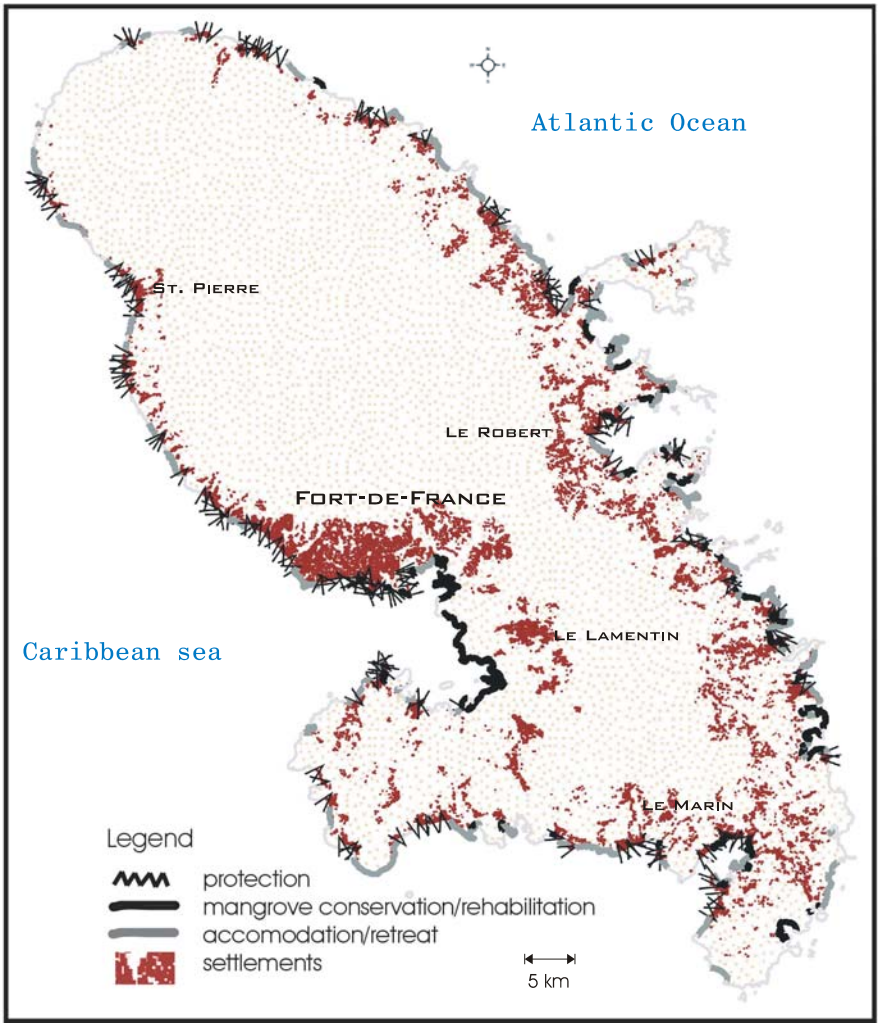


Fig 3.3 potential adaptation measures with respect to vulnerable population to sea level rise impacts.

It is notable that the results of the model differ with each vulnerability factor. An example should make this clear: The optimal adaptation measure of vulnerable population in the Fort-de-France Bay might be the protection of mangrove forests. Concerning the vulnerable infrastructure along this coastal strip the optimal adaptation measure would now partly be protection. The reason is that no humans live within this mangrove area, but the airport of Martinique is situated here. It thus becomes clear that the adaptation measures always rely on the viewpoint of priorities. A combination of all of these single maps into one is not recommendable without knowledge of the regional priorities. Therefore, the maps can be seen as preliminary overview for further local studies.

3.4. Discussion and conclusions

The evaluation of the erosion and inundation risk with rising sea level on Martinique shows a high coastal impact potential. More than 60 % of the human coastal resources are at risk at present conditions and this number will increase if sea level continues to rise. The evaluation of settlements at risk and tourist beaches and accommodations proved very high risk to the majority of buildings and beaches. The main income factor on Martinique is beach tourism (see also PARA ET AL. 2002). Hotels are built close to sea level to facilitate easy access to the beach. If sediment loss further continues, Martinique is at risk of loosing not only the majority of its famous beaches and its valuable mangrove habitats but also its prestige as beach tourist destination. The projected loss of beaches as a consequence of erosion and inundation can cause severe economic impacts on the tourism industry as shown by UYARRA ET AL. (2005). In the mountainous parts of Martinique the small areas adjacent to the beach are often the only flat land available and are therefore intensely used. A retreat back to the hinterland as adaptation to sea level rise is often very complicated for various reasons. One may be the safe distance to the active volcano Mt. Pelée.

Accelerated sea level rise will accentuate the impact and broaden the hazard area. The narrow land adjacent to the beaches is often the only flat land available and densely settled. A retreat back into the hinterland is complicated because of competing land uses, including nature conservation areas of unique flora and fauna and land areas for export agriculture. Furthermore, settlements are prohibited in some areas, for example at the upper slopes of the volcano Mt. Pelée or in river flood plains. The development of a Coastal Zone Management Plan considering sea level rise and its impact area as well as elaboration of public information and evacuation plans is therefore of utmost importance. The best response to sea-level rise and climate change in the coastal zone is therefore an appropriate mixture of adaptation measures (NICHOLLS 2003). The decision of the optimal adaptation strategy depends on the priorities and financial limitations of the responsible authorities. But whatever the final adaptation strategy might be: public participation in decision-making and resource management should be integrated into the planning process.

A study by the World Bank (DEEB 2002) concludes that it is often impossible to conduct vulnerability assessments in the Caribbean because of the lack of adequate data. Also on Martinique no vulnerability assessment has been undertaken, and extreme events like hurricanes are not integrated into the coastal management plan. Besides this data are poor. This study showed that spatial analysis allows the evaluation of potential coastal hazard areas by using an empirical assessment model. The utilization and interpretation of satellite images and other spatial data can partly compensate missing local data. But nevertheless, more background data would improve the accuracy of the vulnerability assessment. Further socio-economic aspects can be easily integrated into the model to illustrate human vulnerability. Through GIS-maps the results are visualised and can be used for public illustration. In this connection the results of this empirical assessment might also serve as base data for more specific economic impact models.

Besides this, the methodology is easily applicable and allows individual transformation to other coasts. As long as adequate data are missing, spatial modelling is a feasible methodology to obtain statements about coastal impacts due to erosion, inundation or sea level rise. It is of importance by localising the hazard areas and for the spatial illustration of human impacts. This GIS analysis gives a spatially explicit assessment of risks that might be further investigated in individual cases. The next step now will be to find a way to put the recommendations into practice and include the findings into a stakeholder dialogue.

Part II

Habitat impact assessments of agricultural land use changes in Eiderstedt

Introduction

Grown up in the region, I reached the age to notice landscape changes on the Eiderstedt peninsula by myself. The drain in whose clear water we have played and fished for sticklebacks a quarter of a century ago has been deepened several times since and its flanks are now steep and only sparsely grown. The intensive fertilization of the arable land but also of the grassland results in reduced water quality. For a long time no fish or frogspawn has been observed there. Landscape changes don't necessarily need to be negative, because quality of life has also improved in the region. But the increasing agricultural intensification with the upheaval of grassland and high fertilizer utilization is followed by serious effects on the landscape function.

The Eiderstedt peninsula may be considered as wetland area: A dense network of drains and former tidal creeks runs through the landscape, additionally the grassland is drained through "Grüppen" – small parallel passing trenches – and the wells for drinking water for the livestock are the artificial water holes within the meadows.

These landscape features make Eiderstedt an important area for meadow breeding birds. Furthermore, the region gets attention each year when large hosts of geese and other migratory birds rest to build up strength on the wet grasslands on their way to or from their wintering areas in the south. Recently, an intense debate was raised about the potential (total or partial) declaration of Eiderstedt into bird sanctuaries within the Natura 2000 network. Both conservationists and local farmers discussed the impact of the declaration with assumptions that are scientifically not scrutinized. A description of the conflict can be found in the following chapters and is also available at www.pro-eiderstedt.de and www.nabu.de.

In this second part we took to the task to examine some aspects of the conflict in more detail. First of all, the analysis of the historical land use changes stood in the foreground combined with the review of the land use conflict between farmers and wildlife. This, combined with socio-economic findings, built the base for the development of future land use scenarios. The central question was a priori, what consequences a projected upheaval of grassland may have for meadow breeding bird species. The opportunity arose to test simultaneously parts of the methodology of the European wetland site-selection model (CHAPTER 8) in a finer scale and to verify the validity of the model through a bird abundance analysis.

4. Agricultural land use changes in Eiderstedt: historic developments and future plans

4.1. Ecological implications of land use choices on Eiderstedt

The peninsula Eiderstedt at the west coast of Schleswig-Holstein is a region that is traditionally mainly used agriculturally. The dominant agricultural land use options are extensive management of grassland and the production of crops on arable farm land. Historically, there have been distinct shifts in the shares of these two land use options, each altering the characteristics of the landscape of Eiderstedt considerably. In times when the focus of agricultural activities on Eiderstedt was on the export of cattle as was the case in the late 19th century (HAMMERICH 1984), practically all agricultural land on Eiderstedt was used as grassland (LVERMA-SH 2007a). But there were also periods in which more than half of the land was arable farm land.

These shifts in land use have ecological implications as Eiderstedt is considered to be one of the prime habitats for meadowbirds in Germany (HÖTKER ET AL. 2005) breeding in the large grassland and wetland areas adjacent to the North Sea. In addition, vast amounts of migrating birds pass through Eiderstedt in spring on their way from wintering grounds in the south to Scandinavia as well as on their way back in fall. The Naturschutzbund Deutschland *(NABU)* classifies Eiderstedt as wetland region of international importance based on the Ramsar convention (NABU 2005). Most of the bird species breeding on Eiderstedt prefer extensively used grassland or wetlands as breeding habitat, while arable farm land is much less suitable for the rearing of offspring.

Currently, approximately three quarters of the agricultural land on Eiderstedt is used as grassland (STAT A NORD 2004). However, plans to increase the share of arable farm land drastically in order to adapt to changes in agricultural production patterns are discussed. Altered boundary conditions brought about by changes in European agricultural policy often necessitate the switch from outdoor dairy production to maintaining the cattle stocks in stables (NEHLS 2002). This means that crops with higher energy content have to be fed, which must grow on arable farm land in the vicinity. These kinds of land use change are

generally irreversible as arable farm land on Eiderstedt needs to be artificially drained so that the original ponds that are characteristic for the landscape in this region are destroyed during the conversion process. According to the local farmers union, two thirds of the agricultural land on Eiderstedt is supposed to be converted to arable farm land within the next couple of decades (NABU 2004). Such a change would not only distinctly alter the appearance of Eiderstedt, but would also mean the loss of valuable bird habitats and possibly a reduction of the recreational attractiveness of the landscape to visitors.

This study will look at possible scenarios of land use development on the Eiderstedt peninsula. After a brief historic overview of past agricultural land use changes in this region, the controversy between farmers and environmentalists about the future development of the local agriculture is presented. Using a geographic information system (*GIS*), scenarios of a future conversion of grassland to arable farm land on Eiderstedt are developed and described. These scenarios can be used in further assessments to quantify the ecological impacts of each development path.

4.2. Historic development of agricultural land use in Eiderstedt

Eiderstedt is a peninsula at the west coast of Schleswig-Holstein that extends into the North Sea. It is located between the river Eider in the south and the town of Husum in the northeast. Back in the 11th century, Eiderstedt consisted of several geest islands, but started to grow together as a consequence of the first coastal protection measures being erected at that time (MEIER 2001). Initially, transportation was only possible by boat as settlements were exclusively accessible from the North Sea. These waterways remained in operation for several centuries and its underlying pattern is still recognizable. Today, almost the entire peninsula is enclosed by dikes built to withstand severe storm floods. This makes it necessary to artificially drain the land area. An extending network of trenches and parallel passing drills (in German: *Grüppen*) on the grasslands have been constructed that have become a typical feature of the Eiderstedt landscape (FISCHER 1997).

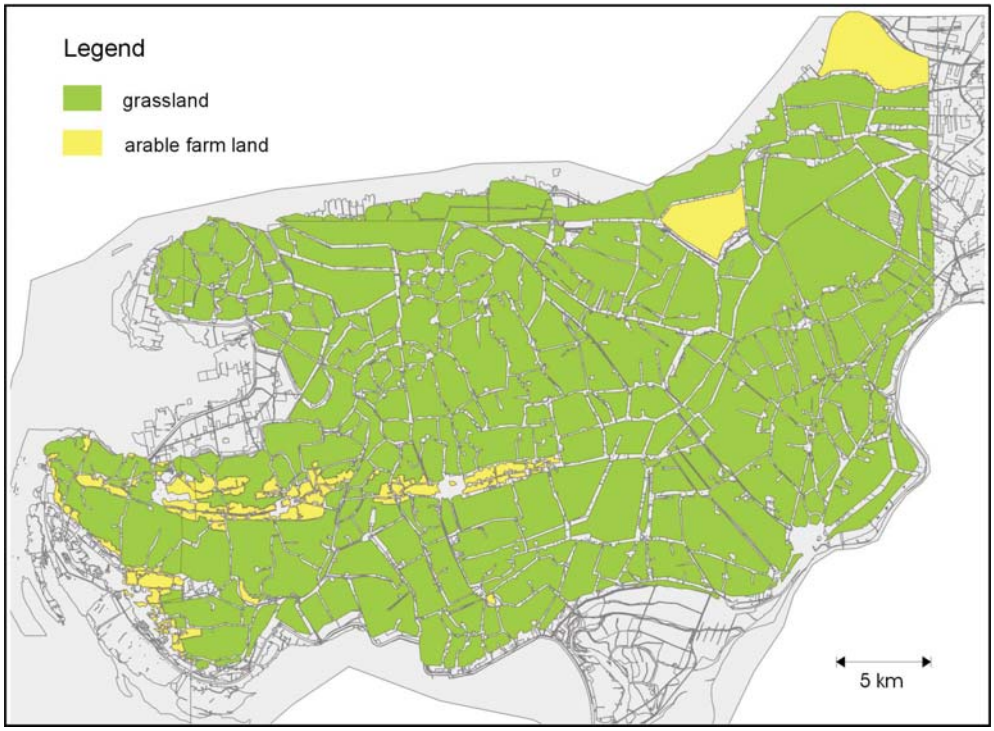


Fig 4.1 Agricultural land use on the Eiderstedt peninsula in 1878, grassland is shown in green and arable farm land in yellow (based on LVermA-SH 2007a).

The soil of the marshland is of high quality (FEDDERSEN 1853; INFONET UMWELT 2007). In the early 19th century crop production was of great importance on Eiderstedt (HAMMERICH 1984) and the share of arable farm land was high. In some years close to half of the agricultural land was used to grow crops. In the middle of that century cattle farming became the prime means of agricultural production as exports of cattle to the United Kingdom via the harbours of Tönning and Husum were very profitable. Consequently, meadows and grassland with ponds and drainage drills running through became the dominant type of agricultural land on Eiderstedt. When detailed maps of Germany were drawn up by the Prussian government in the late 1870s, almost 93% of the agricultural land consisted of grassland (LVERMA-SH 2007a). Arable farm land was hardly found (Fig. 4.1): crop production took place only in the vicinity of the town of Garding and in the northeast of Eiderstedt.

During the first half of the 20th century, there were only little changes in the distribution of agricultural land (HAMMERICH 1984) with the share of grassland always exceeding 80%. After World War II the dairy production became dominant on Eiderstedt, which led to a further reduction of arable farm land

until 1970 (STAT A NORD 1950-2004). Figure 4.2 shows that arable farm land started to increase afterwards, which was mainly due to an expansion of crop production on polders that were secured by dikes in the 1960s.

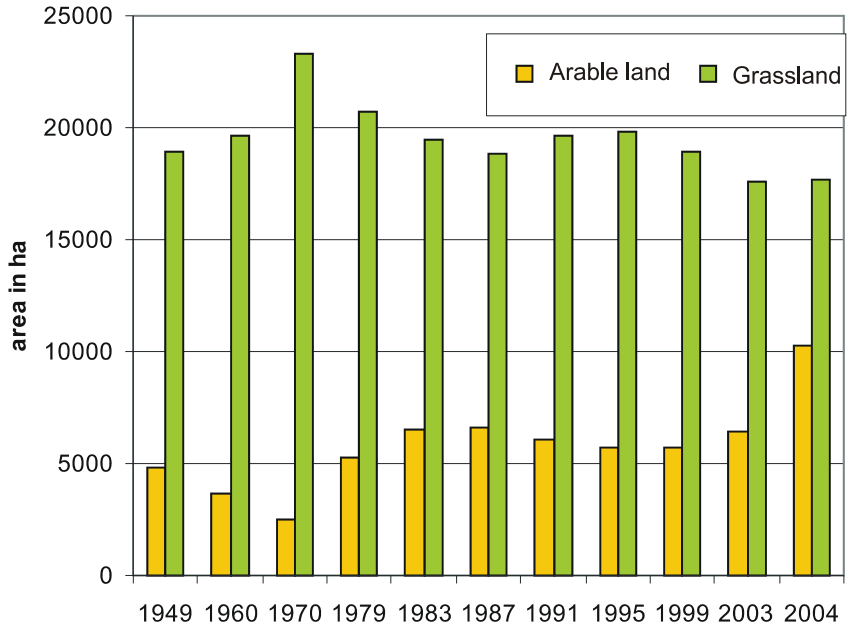


Fig 4.2 Distribution of agricultural land on Eiderstedt: grassland and arable farm land (based on STAT A NORD 1950-2004).

Until 2003 the share of arable farm land remained stable at about one quarter of the total agricultural land. The distribution of the two dominant agricultural land uses in 2002 is illustrated in Figure 4.3, in which the three bird sanctuaries on Eiderstedt (Westerhever, Poppenbüll, and Kotzenbüll) are particularly marked. Even though crops are grown in all regions of Eiderstedt, there are vast areas of contiguous grassland, particularly in central Eiderstedt (LVERMA-SH 2007b). These are of great ornithological significance.

In recent years, however, altered political boundary conditions have caused farmers to switch from dairy production with the extensive grassland use to higher intensity cattle farming and biofuels production. As intensive cattle farming involves permanent housing of the cattle, it is essential to grow the high energy forage crops. The increased demand for these crops and for those used in biofuel production necessitates an expansion of the share of arable farm land at the expense of grassland.

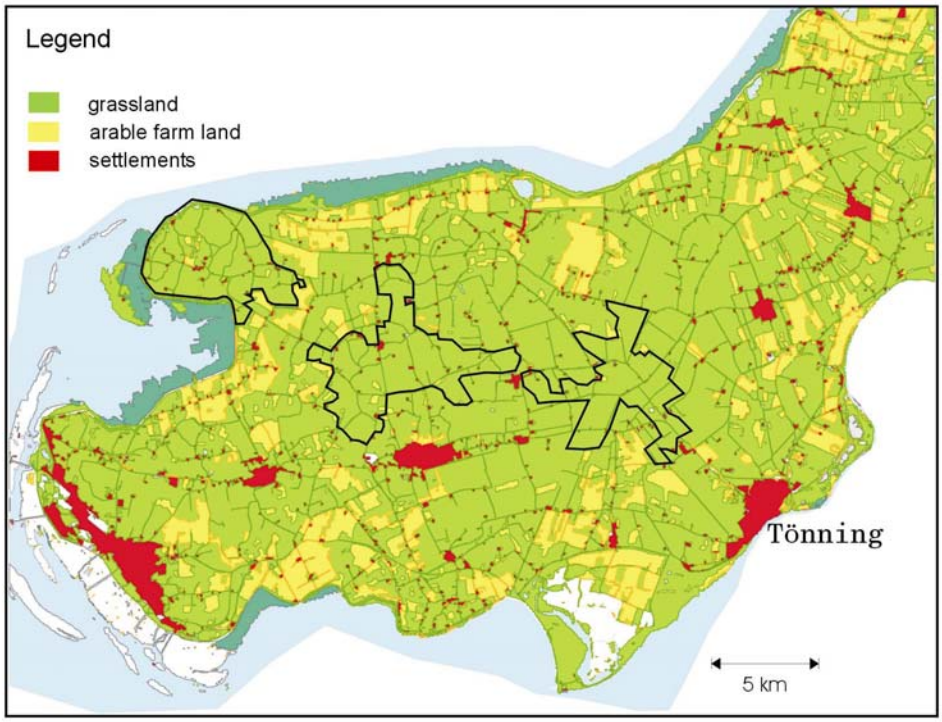


Fig 4.3 Agricultural land use on the Eiderstedt peninsula in 2002 (based on LVERMA-SH 2007b).

4.3. The controversy about plans for future land use change

Grasslands are habitats with the potentially highest biodiversity in central Europe (NEHLS 2002). They are threatened by an intensive agricultural use involving the application of large amounts of fertilizers, the conversion to arable farm land, and dehydration by improving drainage. A conversion of grassland to arable farm land destroys the diverse flora and fauna and cause a deterioration of the quality of the entire ecosystem. The expansion of grassland in other regions to offset the losses is inadequate as newly seeded grassland is ecologically worthless for a long period of time. Consequently, the plan to convert a significant share of the grassland on Eiderstedt to arable farm land is strictly opposed by environmental interest groups led by the *NABU (Naturschutzbund Deutschland).*

Farmers argue that protection plans proposed by *NABU* are far too restrictive and do not fare well with the economic necessities of the region. Their interest group *Pro-Eiderstedt* proposes contractual nature conservation, as conservation

measures can only be realized in consent with the local farmers. Many such contracts were established in the late 1980s but their number declined in the 1990s when fundamental enforcement rules changed. In 2001, approximately 1 000 ha of agricultural land were managed by contractual conservation. According to the Ministry of Agriculture in Schleswig-Holstein, that area increased to about 3 000 ha in 2006 (MLUR 2006). Extensively used grassland managed by contractual conservation may not be converted to arable farm land, drainage may not be intensified, and the application of pesticides and fertilizers is prohibited. Pro-Eiderstedt has developed a concept to manage approximately 10 000 ha of agricultural land by contractual conservation, however, the plan calls for only limited enforcement of the specified rules.

Critics of contractual nature conservation state that it has proved to be not too effective in the past (NEHLS 2002). Contracts with strict rules are hardly attractive to farmers and are therefore very often rejected, even though only one third of such contracts in Germany contain special obligations regarding environmental protection while the largest share of them contains only general rules for extensive land use.

Advocates of strict rules to protect the grassland areas propose to strengthen the extensive grassland use without increasing incentives of a more intensified management. The *NABU* calls for a special premium for the farmers who extensively manage their grassland (NABU 2004) to offset the economic disadvantages of grassland farming in comparison to crop production. An important aspect of this plan is to grant premiums for arable farm land and for grassland separately and with particular reference to the location. Additionally, the premiums must be revoked in case of a conversion of the land. However, the enforcement of such a premium system would be quite complicated and subject to a large number of exceptions.

In addition to economic stimulation, environmental interest groups endorse direct measures to protect ecologically valuable land. The European directive *Natura 2000* requires the members of the EU to identify protected sites according to the European Conservation of Wild Birds Directive. The former environmental minister of Schleswig-Holstein, Klaus Müller of the Green Party, proposed to declare 24 648 ha of Eiderstedt, which is practically the whole area

of the peninsula except for the settlements, as sanctuary. This caused fierce opposition as this plan exceeded the minimum requirements of the directive (SH-LANDTAG 2004). Farmers feared that the declaration of a large protected area would bring them economic disadvantages as new investments and expansions of agricultural activities would be severely regulated.

Instead, three separate bird sanctuaries on Eiderstedt have been declared: one around the town of Westerhever in the north-western corner of Eiderstedt and two others in central Eiderstedt near Poppenbüll and Kotzenbüll (Fig. 4.3). The goal of declaring these sanctuaries was to maintain these sites as habitats for migrating and breeding bird species (MLUR 2006) by preserving the many ponds and drainage drills by limiting the extent of agricultural use. Farmers criticize even this declaration arguing that the EU Conservation of Wild Birds Declaration is only valid for natural and not for cultivated land and does not apply to Eiderstedt as the whole landscape is anthropogenic cultivated in its entirety already for centuries.

Currently, no agreement between the different interest groups appears to be in reach. In case no additional sites are declared as sanctuaries in the future, the remainder of the agricultural land on Eiderstedt may be subject to conversion in the near future. This would alter the appearance of the landscape on Eiderstedt such that arable farm land would become the dominant form of land use for the first time in more than one and a half centuries.

4.4. Scenarios of land use development in the next decades

In order to be able to assess the possible ecological consequences of such land use change, different scenarios are developed. The scenarios are based on the assumption that the plan to drastically increase the amount of arable farm land on Eiderstedt to two thirds of the entire agricultural land area is actually realized within the next couple of decades. Due to the lack of information in the propositions on which areas are to be converted, three different patterns of land use change are compared in the following. The agricultural land use patterns in the course and after the completion of the planned conversion are identified for all scenarios. They differ quite substantially, depending on the development path applied.

The first scenario considers a pattern of land use change, in which land is primarily converted along the main roads through Eiderstedt and preferably in only recently diked marshland (Fig. 4.4).

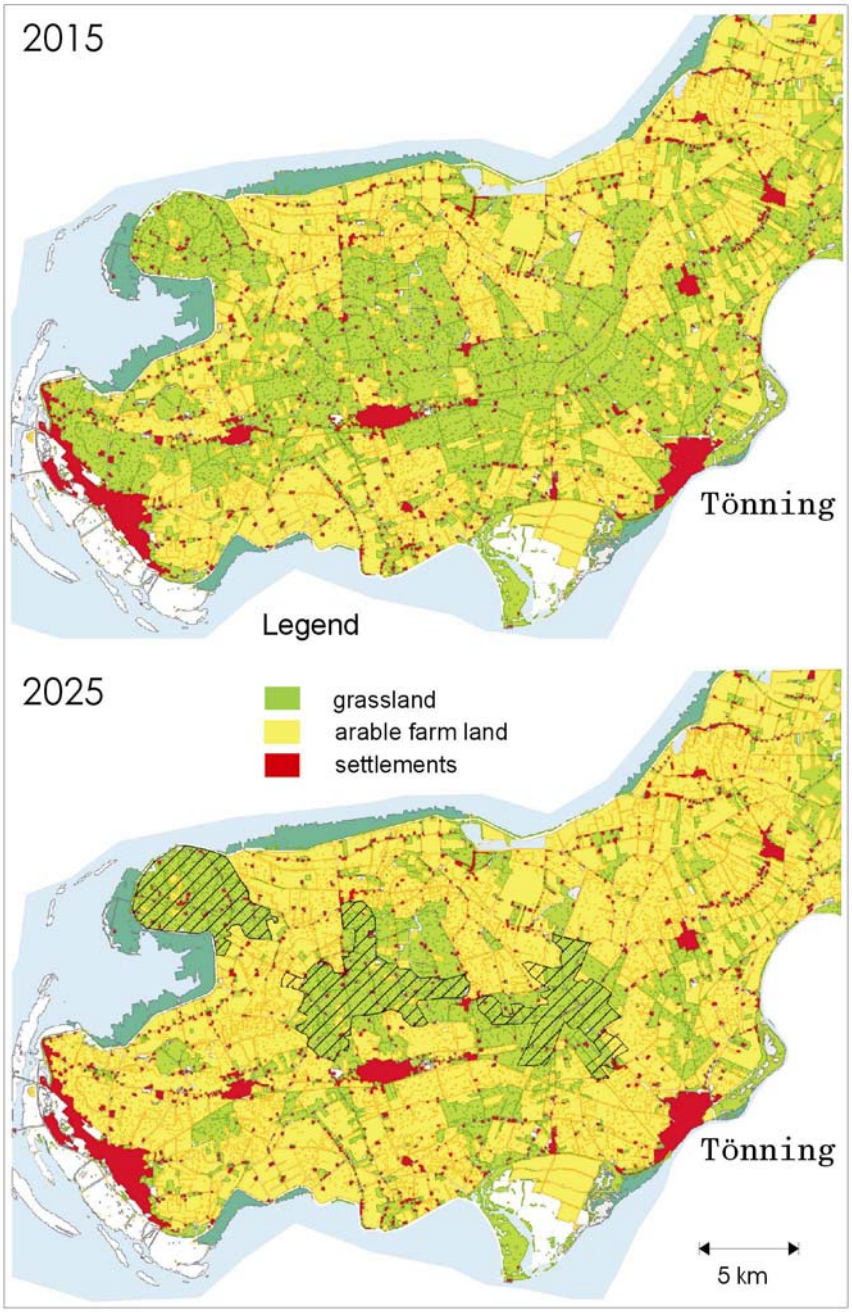


Fig 4.4 Agricultural land use on the Eiderstedt peninsula in 2015 (top panel) and 2025 (bottom panel), if land use change originates along the main roads across Eiderstedt.

Such a development is particularly likely if a lot of biofuels are to be grown on Eiderstedt in the future. Because these crops would need to be transported to the power plants from where they are grown, producing them as closely as possible to already existing infrastructure makes this task significantly easier. If the land use change originates from the main roads through Eiderstedt, the landscape becomes very patchy during the conversion process. Halfway through the conversion process, uniform areas of grassland can only be found in the three declared bird sanctuaries and in their vicinity in central Eiderstedt (Fig. 4.4). The eastern part of Eiderstedt consists of a mix of many small areas of both land uses. It has to be noted that in the early phase of the conversion process land closer to the coastal areas of Eiderstedt are more likely to be converted than the more central parts of the peninsula. The reason is the prioritization of young marshland for arable land. At the end of the conversion process, only some patches of grassland remain scattered throughout Eiderstedt (Fig. 4.4). These are generally quite fragmented, except for the areas around the three bird sanctuaries, in which larger uniform areas of grassland remain intact. These areas would have to serve as primary breeding grounds for the remaining meadowbirds. In all scenarios, the region around Westerhever only remains a uniform grassland area because it is a declared bird sanctuary. If it had not been declared a protected site, the north-western tip of Eiderstedt would also have been converted into arable farm land to a large extent.

The second pattern is based on the assumption that it is best to grow crops on large continuous patches of land. Therefore, in this pattern land is primarily converted in areas around already existing arable farm land (Fig. 4.5). In this case the conversion process is more coherent and produces a less fragmented land use pattern. During this conversion process, a large region of grassland remains in central Eiderstedt. It does not only encompass the two bird sanctuaries but also substantial areas in their vicinity (Fig. 4.5). The large size of this uniform grassland area increases its ecological value as breeding habitat for meadowbirds. Similar to the previous scenario, the arable farm land is mainly located in the regions close to the coast but it is combined into larger units so that crop production can be more efficient in this scenario than in the previous one.

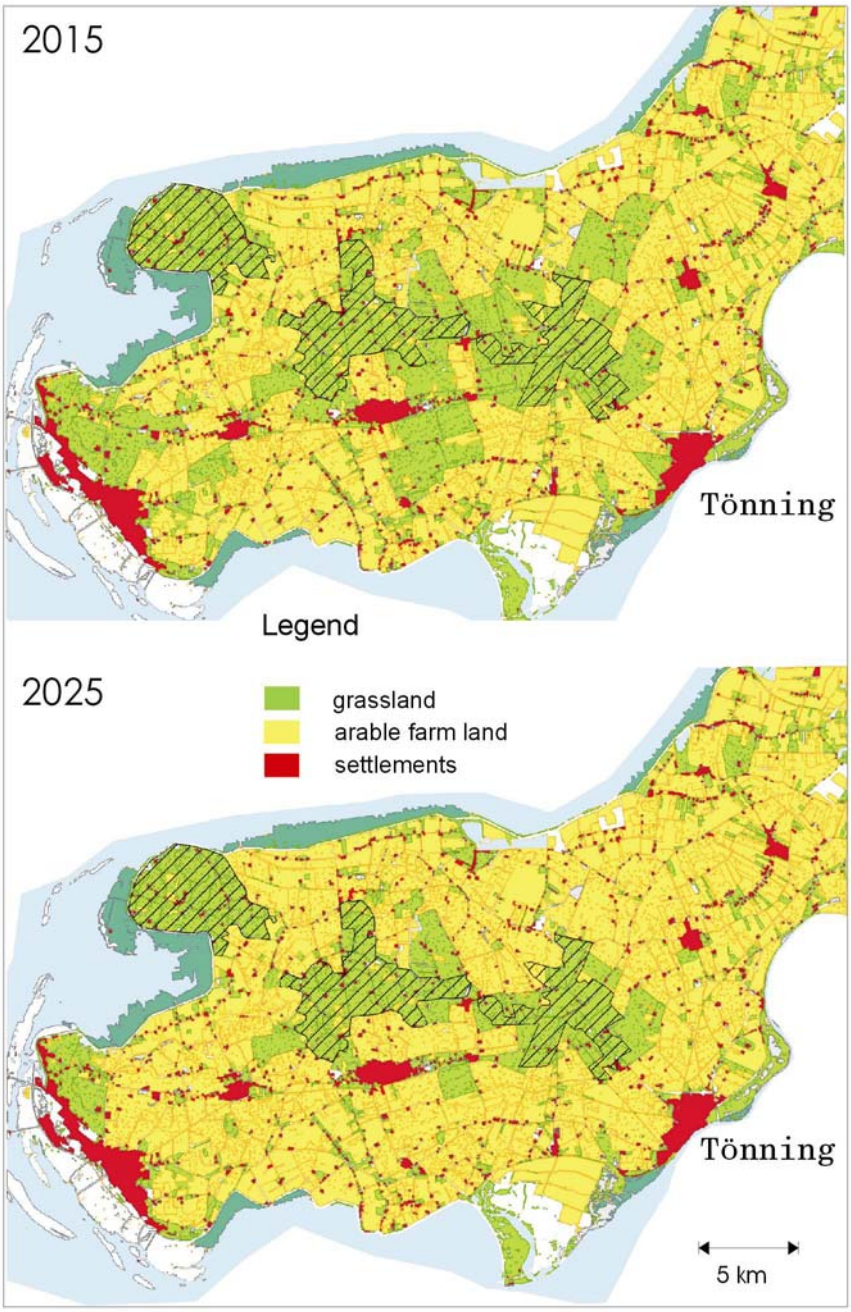


Fig 4.5 Agricultural land use on the Eiderstedt peninsula in 2015 (top panel) and 2025 (bottom panel), if land use change extends outward from already existing patches of arable farm land.

The distribution of the remaining grassland in 2025 in this scenario (Fig. 4.5) is similar to the one considered earlier, except for the region north of St. Peter-Ording, which remains grassland, and the area in central Eiderstedt, where less land is converted in the vicinity of the two bird sanctuaries. In contrast, patches converted to arable farm land are less fragmented, so that the degree of land

use change appears to be even higher than in the previous scenario, even though this is not the case.

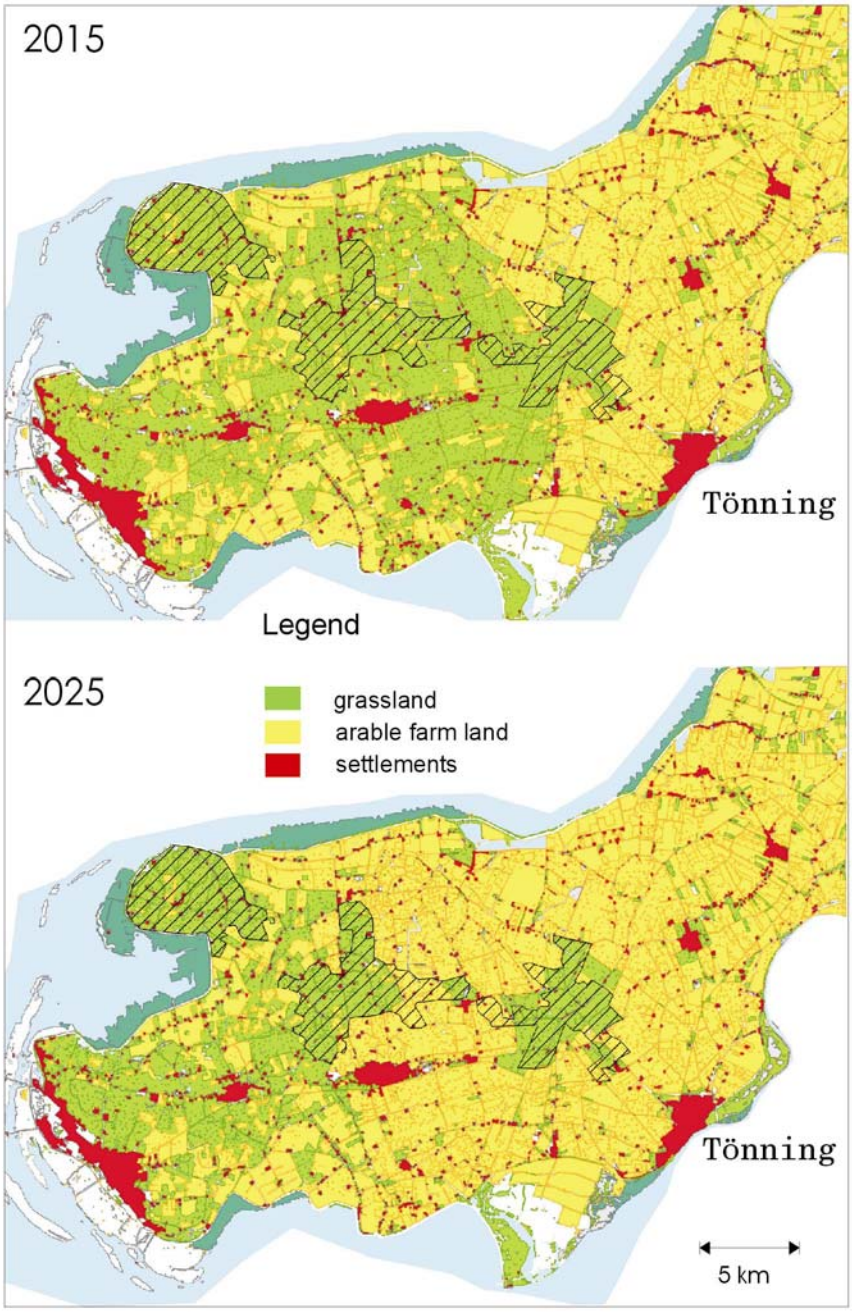


Fig 4.6 Agricultural land use on the Eiderstedt peninsula in 2015 (top panel) and 2025 (bottom panel), if land use change first occurs in the east and then progresses westwards.

The third pattern of conversion follows the premise that the less remote an area of land is, the more useful it is to be used for crop production. Since the Eiderstedt peninsula is connected to the rest of Schleswig-Holstein only in the

east, this pattern of land use change converts grassland to arable farm land from east to west (Fig. 4.6).

By applying such a conversion pattern, it is ensured that the area of arable farm land used for crop production is more or less coherent, while the remaining grassland also consists of large patches as long as possible (Fig. 4.6). In the early phase of land use change, conversions are more likely to occur in the southern part of Eiderstedt than in the north.

If two thirds of the agricultural land have been converted to arable farm land from east to west, practically all of the remaining grassland is confined to the area west of the town of Garding. Eastwards only bird sanctuary of Kotzenbüll remains more or less intact. However, there is even some land use change within the two bird sanctuaries in central Eiderstedt. This is likely to have an adverse influence on the overall ornithological habitat quality of these two special regions.

Furthermore, large parts of the remaining grassland are in the vicinity of the towns of St. Peter-Ording and Tating, which are popular tourist destinations at the west coast of Schleswig-Holstein. This is likely to cause additional ecological difficulties due to increased stress imposed on the fauna caused by high anthropogenic frequentation.

4.5. Possible implications of a conversion of grassland to arable farm land

A large scale conversion of grassland to arable farm land throughout the Eiderstedt peninsula will not only change the appearance of the entire landscape but also have an impact on the number of breeding pairs of meadowbirds supported by the habitats. The scenarios described above are applied in a *GIS* assessment to determine the altered carrying capacity of the Eiderstedt peninsula for key bird species (CHAPTER 5). The results indicate that the pattern of agricultural land use change has a profound influence on how the number of breeding pairs develops.

The three scenarios mainly differ in the location of the remaining grassland areas and in their degree of fragmentation. The fragmentation is highest if the

conversion originates along the existing infrastructure on Eiderstedt, which worsens the quality of the grassland that is not converted to arable farm land. In contrast, the area of coherent regions of agricultural land use is largest if the conversion proceeds westwards across Eiderstedt. However, since the remaining bird habitats in the bird sanctuary near Kotzenbüll become quite isolated and most of the other suitable breeding habitats are located in proximity to major tourist destinations, the bird density in those habitats is likely to decline over time, amplifying the pressure on the bird populations of Eiderstedt caused by the reduction in size of the suitable breeding habitats.

Overall, the quality of the Eiderstedt peninsula as breeding habitat for meadowbirds decreases substantially as a consequence of a large scale land use change. As the density of breeding pairs of four important species declines, the number of individuals supported by the habitats will be reduced at a disproportionately high rate (CHAPTER 5). Even the declaration of the three bird sanctuaries on Eiderstedt will prove to be insufficient to counter this trend since the sanctuaries are also negatively influenced by changes in their vicinity whose influence carries over into the protected areas.

In addition to adverse ornithological impacts, a substantial land use change on Eiderstedt can have an influence on income generated by tourism. Eiderstedt is a famous destination due to its extensive grassland areas and the large numbers of breeding and migrating birds to be observed. The general appearance of Eiderstedt to visitors will change if large parts of grassland are replaced by arable farm land for crop production. How this would influence the tourist industry on the peninsula still needs to be determined separately. The controversy about land use development can only be solved if a compromise can be reached between the ecological demands of the ornithological fauna, the economic interests of farmers, and the aesthetic expectations of tourists visiting the Eiderstedt peninsula.

5. Potential impacts on important bird habitats in Eiderstedt (Schleswig-Holstein) caused by agricultural land use changes

5.1. Introduction

The Eiderstedt peninsula at the Western coast of Schleswig-Holstein (Germany) is a mainly agriculturally used land area which is also home to many bird species breeding along the shores adjacent to the Wadden Sea. Also, vast amounts of birds migrate through this region on their way from their wintering grounds in the south to the Arctic and back. HÖTKER ET AL. (2005) consider Eiderstedt to be one of the most important habitats for meadowbirds in whole Germany.

Currently, most of the agriculturally used land on Eiderstedt is extensively used grassland. In recent years, however, a growing share of the agricultural land is used as arable farm land to grow corn etc. because of an increase in demand for energy-rich food for cattle and fuel for biogas plants that are to be built in the area (HUSUMER NACHRICHTEN 2006). In addition, the extensively used grassland is more and more converted into intensively fertilized meadows for dairy production. Such large scale transformations of agricultural land are likely to have a considerable influence on those bird species that depend on grassland as breeding habitat (BAUER 1997).

In this study, we determine the relationship between the occurrence of birds breeding on Eiderstedt and the characteristics of their breeding habitat. Using this information in a Geographic Information System (*GIS*), we assess the possible impacts of a continued agricultural land use change in the next two decades on the suitability of Eiderstedt as a principal breeding habitat for birds in northern Germany and therefore on the expected abundance of breeding birds in this region.

5.1.1. Agricultural land use changes on Eiderstedt

The Eiderstedt peninsula is located at the west coast of Schleswig-Holstein (Germany). It lies between the river Eider and the city of Husum and extends into the North Sea (Figure 5.1). Until the 11th century Eiderstedt consisted of several smaller geest islands which became connected after the area was enclosed by

dikes (MEIER, 2001). Today, the Eiderstedt coastline is entirely protected by dikes.

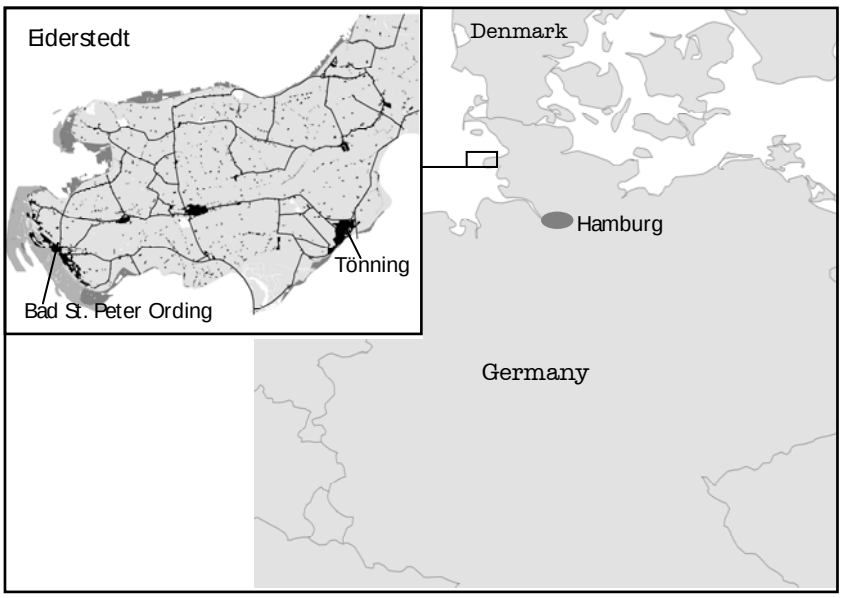


Fig 5.1 The Eiderstedt peninsula.

The soil quality of the marshland is high (FEDDERSEN 1853; INFONET UMWELT 2007). But in order to utilize the land agriculturally, it is necessary to maintain a functioning drainage system. Besides a dense network of narrow trenches between the fields, parallel passing drills (in German: *Grüppen*) that additionally drain the grassland areas are typical for Eiderstedt (FISCHER 1997).

Up to the 18th century, cultivation of crops was one of the prime means of agricultural use of the land, but even though a large share of land was used as arable farm land the predominant type of agricultural land was grassland. Around 1850, cattle farming increased in importance, as exports to the United Kingdom via the harbours of Tönning and Husum grew quickly (HAMMERICH 1984). This called for a considerable extension of opportunities for grazing. In the following decades the share of grassland sometimes even exceeded 90 percent (LVERMA-SH 2007a). After World War II the share of grassland on Eiderstedt decreased from close to 90 percent to approximately 75 percent in the 1970s (Table 5.1) and remained stable at this level until recently (STAT A NORD 1950-2004).

Table 5.1 Agricultural land use on Eiderstedt from World War II until present (STAT A NORD 1950-2004).

year	total agricultural land area (ha)	share of grassland	share of arable farm land
1949	23 691	80%	20%
1960	23 264	84%	16%
1970	25 771	90%	10%
1979	25 973	80%	20%
1983	25 943	75%	25%
1987	25 504	74%	26%
1991	25 698	76%	24%
1995	25 504	78%	22%
1999	24 668	77%	23%
2003	24 016	73%	27%

In recent years, the characteristics of cattle farming on Eiderstedt shifted towards a more intensive approach with the cattle for meat production remaining in cow barns while the cattle used in dairy production is held on grassland, which is often heavily fertilized. The increased number of cattle held necessitates the growth of large amounts of forage crops in adjacent areas, mainly corn. Since the total agricultural land area on Eiderstedt is limited, this led to a considerable increase in the share of arable farm land on Eiderstedt since 2003 and a concurrent reduction of grassland. Adventitiously, enhanced grassland conversion takes place because of fuel production for biogas plants.

The local farmers union plans to extend the amount of land used to grow forage crops to approximately two thirds of the agricultural land area in the next couple of decades (NABU 2004). This plan is intensely debated and opposed by local environmental organizations who claim that such a large scale shift in land use not only alters the overall appearance of the whole region but also has devastating effects on the breeding bird colonies, as arable farm land on which corn is grown is much less suitable as habitat than extensively used grassland (BEINTEMA 1983).

Therefore, realization of the farmers' plans would greatly impact the local carrying capacity of many (endangered) bird species.

5.1.2. Eiderstedt as important habitat for breeding and migrating bird species

Grassland is often an important substitute for lost natural habitats such as moors, salt marshes, or other wetlands. Eiderstedt offers ideal breeding conditions for meadowbirds owing to its large share of grassland and meadows with many ponds and drainage trenches that are extensively used by the local agriculture.

The Eiderstedt peninsula is an important breeding area of the Northern Lapwing (*Vanellus vanellus*), the Eurasian Oystercatcher (*Haematopus ostralegus*), the Black-tailed Godwit (*Limosa limosa*), and the Common Redshank (*Tringa totanus*) in Germany (HÖTKER ET AL. 2005). Despite considerable measures to protect the populations of these species, their abundance has decreased dramatically during the last few years. Northern Lapwing and Common Redshank are considered to be endangered; the Black-tailed Godwit is even listed in the category of being severely threatened by extinction (BAUER ET AL. 2002; KNIEF ET AL. 1995). On the other hand, the abundance of the Eurasian Oystercatcher has increased recently.

The selected birds depend on low and sketchy vegetation on wet meadows or marshes (GILLMOR ET AL. 1998). Some prefer the proximity to open waters but all avoid fallow lands and cut meadows (HOFFMANN 2006). The selected birds can also serve as indicators in land use intensity assessments (BEINTEMA 1983). While Eurasian Oystercatchers and Northern Lapwing are also found on intensively used grassland and sometimes even breed on arable farm land, Common Redshank and particularly Black-tailed Godwit have higher demands regarding the management intensity of the grassland (BEINTEMA 1983; HOFFMANN 2006).

5.2. Methodology

5.2.1. Aims and methods

Conversion or the abandonment of extensively used grassland to either arable land, intensively used grassland or to fallow land with forest succession can be observed throughout Europe. Such land use changes are often motivated by

political decisions and demographic or socio-economic trends (BAUER ET AL. 2002). Regardless of the reason, a loss of valuable habitat can generally be registered as a consequence of such land conversion (EEA 2004). This has implications for the ornithological fauna, which manifest themselves in the fact that many farmland bird species have been declared endangered species in Europe over the last few decades and that their decline has become an important conservation concern (BAYLISS ET AL. 2005). This study aims to improve the understanding of the potential impacts of land use changes on key species of the local bird fauna by exploring a set of possible land use development scenarios. We focus on four bird species with mapped field distribution as key species. The following key questions serve as guideline for the assessment:

1. Which processes cause the land use change and how can these be transformed into future land use scenarios?

2. How are the breeding habitats of the key species characterized and how can these be assessed concerning its site and habitat suitability?

3. To what extent do the habitats change qualitatively and quantitatively with the land use change scenarios?

4. What implications does this have on the key species?

5. Can general statements to grassland conversion be deducted from the findings?

Several empirical models already exist, which can be used to analyze the distribution and habitat suitability of species. Most of them model the potential distribution of certain single or multiple species (MANEL 1999; CABEZA ET AL. 2004; SEOANE ET AL. 2004; BAYLISS ET AL. 2005). BAYLISS ET AL (2005) use a multi-species approach that utilize Bayesian decision rules, others like SEOANE ET AL. (2004) apply predictive habitat models or general linear models (GUISAN 2002; GRANADEIRO 2004). In our assessment, the emphasis lies on the utilization of existing field data of bird occurrence and their extrapolation in accordance with different land use change scenarios. The analysis is conducted with a GIS-based model that is explained in more detail below.

GIS methods have already been used in some studies to determine species distributions. E.g., THOMPSON ET AL. (2004) identify locations of potential breeding sites of curlews and POWELL ET AL. (2005) analyze species distributions

using biotic and abiotic factors to predict former ranges of species. They also demonstrate that simple rule-based non-statistical models can be effective tools for such applications. However, the integration of scenarios into GIS-based modelling of species habitats has been often neglected so far. This necessary step forward, which allows the application of the results in effective land use planning and conservation management, is taken in this study.

5.2.2. Data and software used in the assessment

In order to be able to determine the potential impacts of future land use changes on the breeding populations of the bird species in question, it is necessary to look at the historic land use development as it defines the current situation on Eiderstedt. This is done using survey data on agricultural production in Schleswig-Holstein provided by the Statistics Department Nord, which allows us to assess the period from the end of World War II until present (STAT A NORD 1950-2004). Together with GIS data on current land use on Eiderstedt, provided by the Landesamt für Natur und Umwelt des Landes Schleswig-Holstein (LVERMA-SH 2007b) and data on the abundance of key bird species breeding in the area (HÖTKER ET AL. 2005) it is possible to relate the preferred breeding habitats to agricultural land use decisions. ArcGIS as well as the analysis tools V-late and Hawths Analysis Tools (2006; TIEDE 2005) are used in this assessment.

The development of agricultural land use in recent decades is extended into the future for another 20 years. The projections are based on political intentions to drastically increase the share of arable farm land up to two thirds of all agricultural land on Eiderstedt (NABU 2004) and assumptions about the possible patterns of land use change. As these changes alter the suitability of the land to serve as breeding habitat for meadowbirds, they can be expected to have a profound influence on the number of breeding pairs on the peninsula. The extent of the ornithological impact is quantified using a measure of dynamic habitat sensitivity of the potential breeding areas.

5.2.3. The Habitat-Sensitivity-Index as measure of biotope quality changes

We developed an assessment scheme to determine how landscape changes affect the characteristics of breeding habitats of birds. This scheme includes the transformation of ecological facts, effects and connections into indices that can be

used in an objective interpretation (see BASTIAN & SCHREIBER 1999; WEIS 2008). The habitat assessment relies on a combination of specific algorithms that allow integrative and complex statements (BASTIAN 1997). Together with the results of the scenario analysis, these statements are projected into future conditions and compared with each other. This allows assessments of the impact potential of land use change and the sensitivity of the landscape. The following equation provides the basis of the habitat assessment. The habitat sensitivity (*HaSI*) of each patch of land *i* is a combined measure of three key indices: the proximity index (*PX*), the neighbourhood quality index (*NI*), and the patch size index (*SCI*). *DI* denotes the habitat demand index.

$$HaSI_i = \frac{\sum_i \left(PX_i, NI_i, SCI_i \right)}{3} \forall DI \in [4,5] \quad (1)$$

A fundamental element influencing habitat sensitivity is the development of the suitability of land as ornithological habitat. It is described by a habitat demand index (*DI*). This index measures the suitability of a number of land cover parameters for selected breeding birds. Sites that have a relatively unfavourable natural character but serve as habitat for the majority of breeding birds can receive a fairly high index value as well. The supply (of nature) is related to the demand of the potential user (in this case endangered bird species).

The analysis of the habitat demand of selected bird species is based on the procedure of a habitat suitability analysis conducted by LANG & BLASCHKE (2007). The data used here are adapted from occurrence maps of selected breeding birds of 2001. Spatial land use and biotope data are from 1991 and 2002. In our analysis, we determine the preferred habitats of the selected bird species. In a first step, the occurrence data are intersected with the biotope and land use maps to identify the preferred habitat types. The results, which are expressed as proportional shares, are subsequently transformed into ordinal classes with five categories, the *DI*. Because we use data on breeding birds only, it is necessary to supplement the data with further information from literature (MORRISON ET AL. 1992; GILLMOR ET AL. 1998; GRUBER 2006; HOFFMANN 2006) to avoid uncertainty errors as described in LANG & BLASCHKE (2007). The *DI* categorizes the degree of general habitat suitability, which is the basis for further analyses.

The resulting list of suitable and therefore valuable habitats for the selected bird species is space independent and yields information on a functional level. Biotope types with a *DI* of 4 and 5 (60–100%) are considered to be potentially extraordinary or very suitable habitats, whereas a *DI* of 3 (40–60%) refers to conditionally suitable or partially suitable habitats. *DI*s of 2 and 1 (0–40%) are unsuitable as habitats for the selected bird species and are omitted in the following model analysis.

The results gained above are spatially transformed to fit the biotope data of Eiderstedt and the areas with high habitat suitability, i.e. a *DI* of 4 or 5, are selected for further analysis. These particularly suitable habitats are the basis for the isolation assessment via the proximity index (*PX*) (GUSTAFSON & PARKER 1992). The proximity evaluation is conducted with the tool V-late (LANG & TIEDE 2003) for *ArcGIS* and allows the rating of individual patches of land according to its functional network with the surrounding habitats (KIEL & ALBRECHT 2004). The *PX* distinguishes between space dispersal and clustered distribution of habitats by considering the size as well as the distance of the patches. Both quantities are important for the assessment of habitat complexes. We use 2002 as base year with a buffer of 250 m for the *PX* evaluation. The results are transformed logarithmically and split into five classes (based on WEIS 2008). For comparability, the same divisions are applied in the subsequent scenario analyses. The index decreases the smaller the area and/or the higher the distance to similar patches of land becomes. The index value is highest if a patch is surrounded by and/or extending towards nearby biotopes of the same kind (LANG & BLASCHKE 2007). Table 5.2 shows the classification scheme of the *PX*.

Another important aspect in the evaluation of habitat sensitivity is the character of the environment (BASTIAN 1997), since it also plays a role in the habitat choice of the bird species (NEWTON 2003). In our assessment this is denoted by the neighbourhood index (*NI*).). It is assumed that the NI, and therefore the attractiveness of the area for the selected bird species, declines with a diminishing quality of the surrounding environment. We follow the assessment of WEIS (2008). The *NI* can only be calculated if the following information on habitat quality is given.

The quality of the neighbourhood is determined by the *BQ* index (cf. SCHLÜTER 1987). The *BQ* consists of an assessment of all characteristics of an area under utilization-specific aspects. This index value is represented by a combination of the hemeroby index (*HI*) and an index of the conservation value (*CI*).

$$BQ = \frac{HI + CI}{2} \qquad (2)$$

Hemeroby is based on vegetation and depends directly on human utilization intensity and pressure. It assesses how pristine a considered biotope is, given the influence of anthropogenic cultivation present. SCHLÜTER (1987) developed a scale to rate biotopes based on their vegetation. For our purposes, this scale is adjusted to yield five index classes by aggregating two levels into one index class. Generally, open water do not fit into this scheme. However, the North Sea and the river Eider are included in our assessment and rated as *HI* = 5 because these waters are of significance for the adjacent salt marshes and the bird species considered in this study. The index values for the open water are also important to prevent a bias in the classification of the *NI*. The *HI* is closely connected to the biological regulation and regeneration capacity. The lower the *HI*, the more limited the regulation and regeneration potential of the biotope is. This allows inferences about the ecological stability of assessed landscapes.

In addition to the *HI*, the *CI* is a second measure of biotope quality. Each biotope is evaluated according to its general importance for species and biotope conservation. We apply the assessment scheme presented in BASTIAN & SCHREIBER (1999) which is adapted to fit our model. The base data are provided by a biotope map of 1991 (LANL 1993) that is classified in a GIS. As before, index values between 1 and 5 are assigned to each biotope type based on its general conservation value. Table 5.2 lists the classification schemes of habitat quality for each biotope type. The characteristics of all index classes are described in table 5.3.

Table 5.2 Index classes of different biotope types on Eiderstedt used in the analyses.

index	*PX*	*BQ*	*NI*	*SCI (ha)*
1	0.000 - 1.231	roads, other paved areas	0.0 - 2.0	0.0 - 2.0
2	1.232 - 2.569	settlements, arable farm land	2.1 - 2.5	2.1 - 10.0
3	2.570 - 3.453	intensively used grassland	2.6 - 3.2	10.1 - 40.0
4	3.454 - 4.211	extensively used grassland, beaches, dunes, ponds	3.3. - 4.0	40.1 - 100.0
5	4.212 - 6.078	marsh land, salt marshes, forests, open water	4.1. - 5.0	> 100.0

To obtain the *NI*, a buffer of 250 m is applied to all areas. The area-relevant mean value of *BQ* is determined for each patch (BASTIAN 1997). The difference between the *BQ* of the habitat and that of its surrounding is a measure of the quality of the neighbourhood (WEIS 2008). It is expressed in five index classes.

Table 5.3 Description of index values.

index	***PX***	***NI***	***HaSI***
1	isolation of small habitat patch is extremely high	quality of the surrounding area is extremely unfavourable	extremely low bird habitat quality. No ecological value for selected bird species.
2	high isolation or very small habitat patch	neighbouring areas of worse ecological quality	low habitat quality with minor value for selected breeding birds
3	medium isolation or medium sized habitat patches	medium neighbouring quality	medium habitat quality but still of value for selected bird species
4	habitat patches build small complexes or are of bigger size	good biotope quality of the neighbourhood	good habitat quality with significant value for selected breeding birds
5	very high complexity or extending patch size	excellent biotope quality of the surroundings	excellent habitat quality with very high ecological value for selected breeding birds

The size of a patch is also of importance when considering its neighbourhood. The larger the habitat the less is it influenced by its surroundings. To integrate the habitat size, the area of each habitat is determined and transformed into five area size classes, the *SCI* (Table 5.2).

The *HaSI* integrates all elements described above and allows an assessment of the state of the landscape with special consideration of the necessities of the selected breeding bird species. The *HaSI* is first determined for the base year 2002, which is the reference for the comparisons with the different scenarios of land use development on Eiderstedt.

5.2.4. Implications for bird populations

Occurrence maps of the selected bird species are intersected with the *HaSI* index values to determine the mean breeding pair density for each *HaSI* class (Table 5.5). Because the bird data only maps occurrence within the dikes the outer salt marshes are excluded from further analyses. It is assumed that these areas, which often have the highest abundance of breeding birds, remain stable in size and carrying capacity. Under the condition that bird abundance per unit area is time independent, we incorporated the results into the scenarios, allowing calculations of the potential reduction of breeding pairs of the selected species. This assessment is conducted for each single bird species separately but also for all four species densities taken together. In the appendix 5.6 the spatial distribution of bird densities is illustrated in more detail.

5.3. Scenarios

In our assessment we assume that the plan to drastically increase the amount of arable farm land on Eiderstedt is realized within the next couple of decades. Since the propositions do not contain any information on which areas are to be converted, three different patterns of land use change are considered in this analysis. They are shown in figure 5.2 together with the current agricultural land use on Eiderstedt (Fig. 5.2a). In the first pattern of land use change, land is primarily converted along the main roads through Eiderstedt and preferably in only recently diked marshland (Fig. 5.2b), as the crops to be grown on the converted land need to be transported efficiently to the sites at which they are

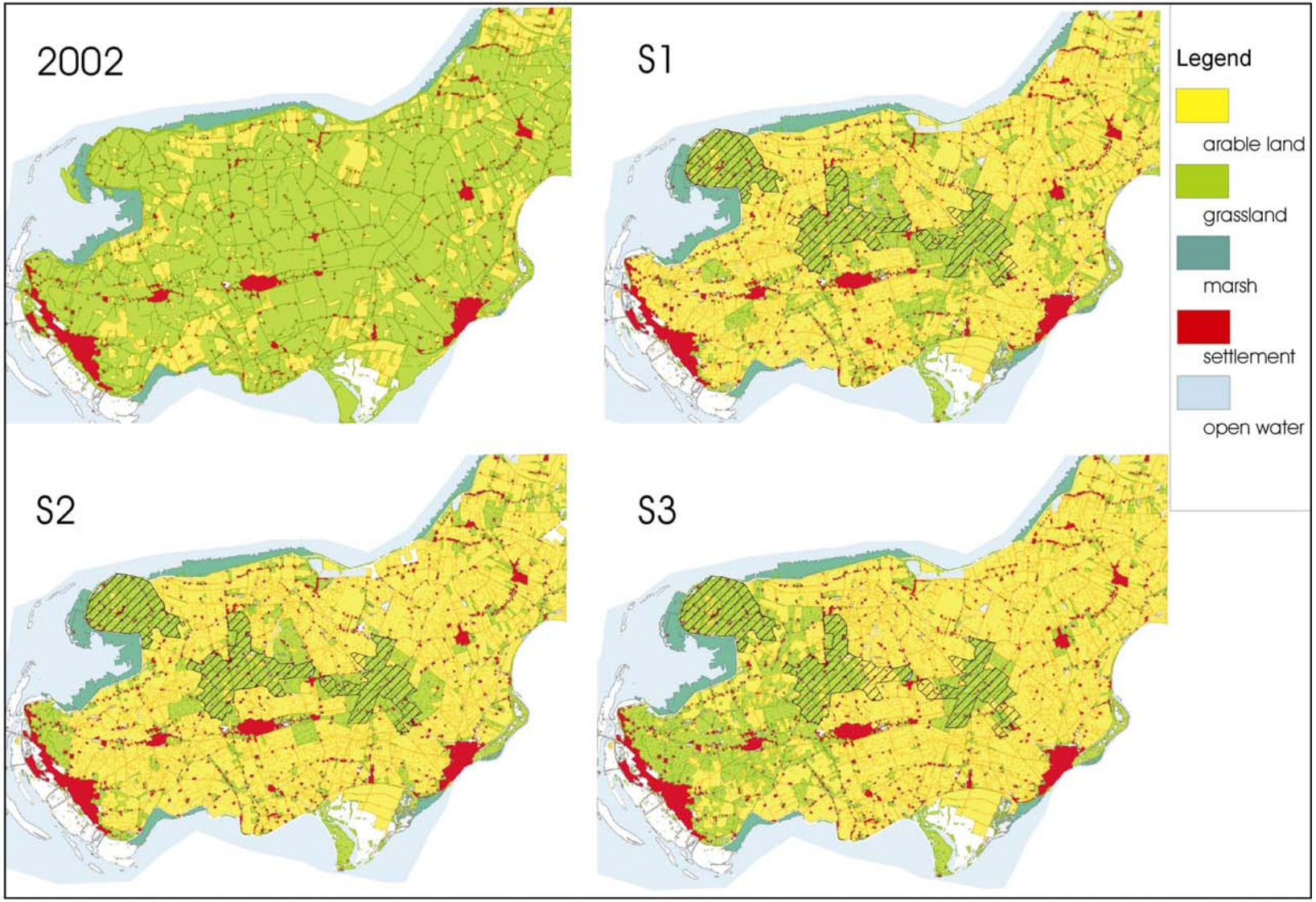


Fig 5.2 a) land use on Eiderstedt in 2002; expected land use on Eiderstedt in the late 2020s if land use change occurs b) along main roads and newly diked areas (*S1*), c) around already existing arable farm land (*S2*), d) from East to West (*S3*).

processed. Growing the crops as closely as possible to already existing infrastructure makes this task significantly easier. The second pattern is based on the assumption that it is best to grow crops on large continuous patches of land. Therefore, in this pattern land is primarily converted in areas around currently existing arable farm land (Fig. 5.2c). The third pattern of conversion follows the premise that the less remote an area of land is, the more useful it is to be used for crop production. Since the Eiderstedt peninsula is connected to the rest of Schleswig-Holstein only in the East, this pattern of land use change converts grassland to arable farm land from East to West (Fig. 5.2d). Further details about these scenarios are given in CHAPTER 4.

5.4. Results

The agricultural land use patterns resulting from the planned conversion are identified for all scenarios. Afterwards, possible impacts on the populations attempting to breed on Eiderstedt are determined by considering the previously obtained information on the breeding habitat preferences of the bird species assessed.

Depending on the pattern of land use change, the scenarios lead to considerably different distributions of agricultural areas on Eiderstedt approximately two decades into the future. If the land use change originates from the main roads through Eiderstedt (scenario *S1*), patches of grassland remain throughout Eiderstedt (Fig. 5.2b). These are generally detached from one another, except for the areas around the three bird sanctuaries, in which larger uniform areas of grassland remain intact. These serve as primary breeding grounds for the remaining meadowbirds. It has to be noted that the political choice of declaring Westerhever a bird sanctuary is the only reason for not converting the north-western tip of Eiderstedt into arable farm land.

The distribution of the remaining grassland in 2025 is similar in the scenario *S2*, in which land use change radiates outward from already existing patches of arable farm land (Fig. 5.2c). The region north of St. Peter-Ording remains grassland and less land is converted in the vicinity of the two bird sanctuaries in central Eiderstedt. Patches converted to arable farm land are less fragmented, so that the

degree of land use change appears to be even higher than in the previous scenario, even though this is not the case.

The resulting pattern is substantially different in scenario *S3*, which depicts a conversion of farm land progressing westwards (Fig. 5.2d). Practically all remaining grassland is located west of the town of Garding. Eastwards only bird sanctuary of Kotzenbüll remains more or less intact, even though it has to be noted that there is even some land use change within the two sanctuaries in central Eiderstedt, which is likely to have an adverse influence on the overall habitat quality of these two special regions. Another caveat is that large parts of the remaining grassland are in the vicinity of the towns of St. Peter-Ording and Tating, which are popular tourist destinations at the west coast of Schleswig-Holstein. High frequentation of the areas surrounding the breeding habitats by humans can artificially reduce breeding success even though habitat conditions might be superior to those in the other two scenarios.

In the next step of the assessment, each patch of land is characterized based on the classification criteria outlined above. This way it can be determined how the altered land use patterns in all scenarios influence the suitability of the land as breeding habitat for the various bird species.

The proximity index yields information about the isolation or complexity of habitats. The analysis reveals that in 2002 areas with a high *PX*, i.e. a high complexity of habitats, are evenly distributed across the peninsula. Only very small patches and adjacent salt marshes have lower index values. The results of the neighbourhood quality evaluation show the same pattern except that the salt marshes now have highest index values. In contrast to 2002, the index values are much lower in all three scenarios, but there are clear differences between the three cases considered. It is striking that the values for the salt marshes remain unchanged with the exception of the *NI* in *S2*, in which they suffer from extremely reduced biotope quality in neighbouring biotopes.

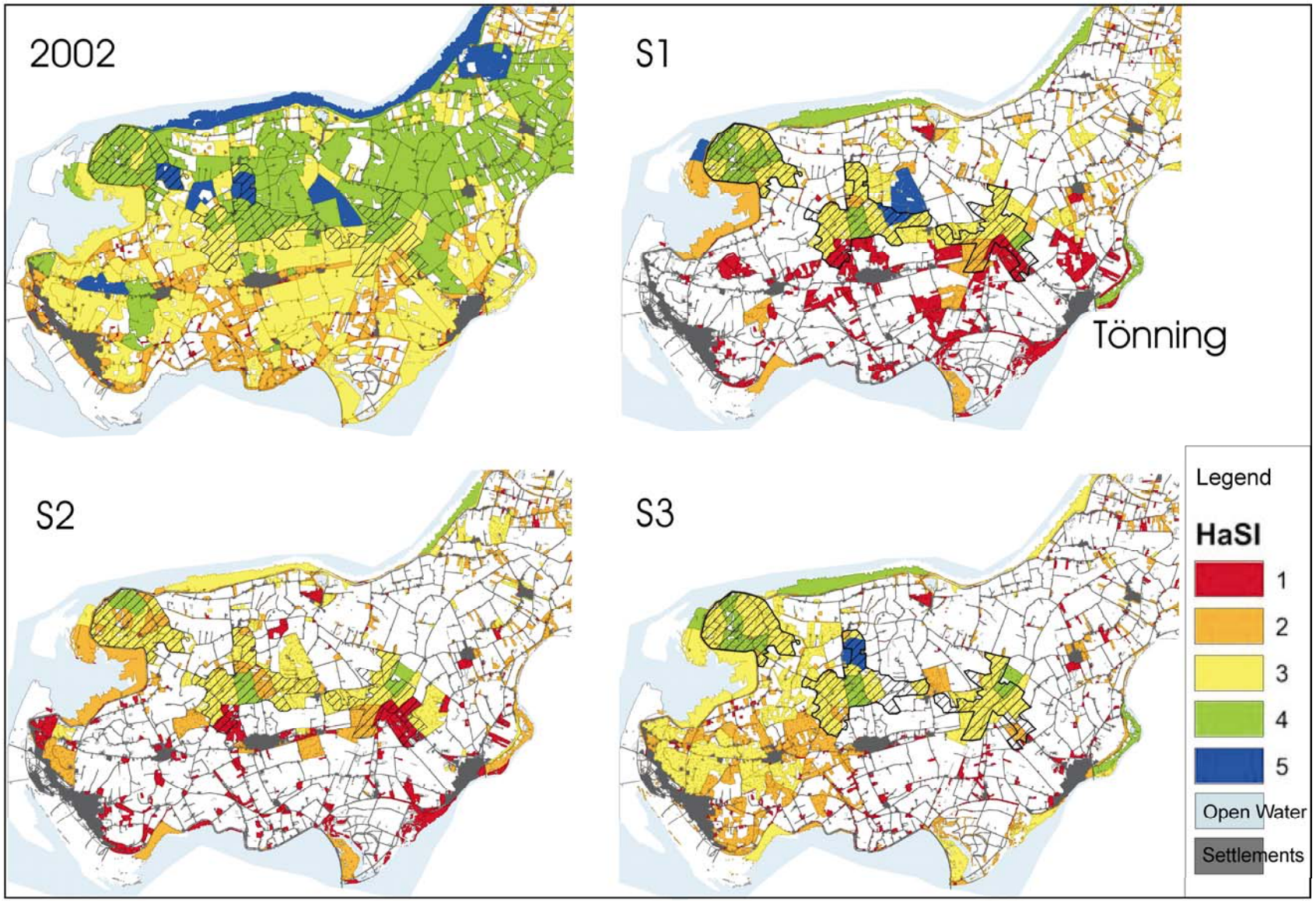


Fig 5.3 The *HaSI* for a) 2002, b) scenario *S1*, c) scenario *S2*, d) scenario *S3*.

After integrating all intermediate results in the *HaSI* equation, it is possible to draw conclusions about changes in habitat quality. *HaSI* ranges from 1 to 5, with class 1 referring to the lowest possible habitat quality with almost no ecological value for the selected breeding birds. The *HaSI* for 2002 and for the three scenarios is illustrated in figure 5.3. In 2002, 26 132 ha of valuable habitats for the selected birds, expressed by a *DI* of 4 or 5, have been available. This amounts to 70% of the total land area of Eiderstedt. The habitats are evenly distributed throughout the peninsula. Seven patches of land are rated with the highest *HaSI* value of 5. These are the salt marshes along the northern coast, as well as patches situated in the northern half of Eiderstedt. Only one patch is located in the south-western part close to St. Peter-Ording. The habitats in the northern and eastern part of the peninsula obtained mainly high *HaSI* values of 4, whereas the lower values of 2 and sometimes of 1 are generally found in the south.

The three scenarios of possible development of agricultural land use are now compared to the reference state of 2002. Besides the reduction of total suitable habitat area, changes in *HaSI* of the remaining suitable breeding habitats are evident. Only in scenario *S3* one of the former patches with an index value of 5 remains, all others are either converted into arable farm land or have a deteriorated *HaSI*. In *S1*, the most suitable habitat shifts towards the centre of Eiderstedt. In *S2*, areas with the highest *HaSI* no longer exist and also the second highest index class is only found in four areas. Table 5.4 shows the share of the area for each *HaSI* class.

Table 5.4 Shares of land area (%) in each *HaSI* class in the three scenarios and the reference year.

HaSI	2002	*S1*	*S2*	*S3*
1	0.4	30.1	23.6	6.9
2	11.1	18.2	28.4	30.2
3	41.8	37.4	42.2	52.5
4	41.5	9.9	5.9	8.6
5	5.1	4.4	0	1.8

It highlights the differences between all three scenarios but also gives the overall proportional changes in habitat quality. The area of habitats with lowest *HaSI* increases considerably in all scenarios. While in 2002 only 0.4% of the area is rated with *HaSI* of 1, the amount of land in this category increases to 7% in *S3*. In *S1* it even reaches nearly one third of the demanded area. The most dramatic change occurs with land that is fairly well suited as breeding habitat (*HaSI* = 4): In 2002, 41.5% of the total area represents this habitat quality, whereas after the presumed land use change only 6 to 10% of the remaining area are still well suited as breeding habitat. To sum up, in all three scenarios large amounts of previously highly suitable habitats are degraded to sites with medium or low habitat value.

The results of the habitat sensitivity analysis are used to obtain average breeding pair densities of the selected bird species for each *HaSI* class. First of all, the breeding pair density is determined for each bird species and in total for the base year 2002. The breeding pair density of all four species considered is positively correlated with the *HaSI* (Tab. 5.4). This is a very convenient finding, as it also verifies the methodology of the *HaSI* evaluation. Only few Eurasian Oystercatchers breed on patches with a poor *HaSI*, while all other birds prefer higher quality habitats. Assuming that the bird densities remain stable for each *HaSI* category, it is possible to calculate the potential abundance of breeding birds in each scenario. Table 5.5 gives an overview of the breeding pair density per HaSI and the resulting bird abundance in the scenarios. The scenarios point to considerable impacts on breeding habitats caused by large scale agricultural land use changes: There is not only a loss of total habitat area of approximately two thirds, but also a shift in the quality of the habitats. With bird densities remaining constant over time, a decrease in bird abundance of more than 60% can be expected. Compared to the 32 895 breeding pairs of all four bird species in total in 2002, the number of pairs should decrease to about 11 000 pairs.

The actually determined expected number of breeding pairs shown in table 5.5 is even lower since the reduction in suitable land area brings about a decline in quality of the remaining habitats. The numbers of breeding pairs decline by another 50% in some scenarios due to this additional effect.

Table 5.5 The average breeding pair density for each habitat sensitivity index (*HaSI*) class (breeding pairs/ha) and the total expected bird abundance in all scenarios.

bird species	*HaSI*	Density	2002	*S1*	*S2*	*S3*
all species	1	0.04	4	87	62	20
	2	0.47	1164	619	877	1042
	3	1.21	11235	3272	3358	4179
	4	1.87	17252	1343	720	1180
	5	2.84	3240	903	0	372
	total		32895	6224	5017	6793
Eurasian Oystercatcher	1	0.03	3	64	47	15
	2	0.12	292	158	224	266
	3	0.31	2973	838	860	1195
	4	0.52	4804	373	200	328
	5	0.65	740	207	0	85
	total		8812	1640	1331	1889
Common Redshank	1	0	0	0	0	0
	2	0.02	58	26	37	44
	3	0.06	564	162	167	231
	4	0.13	1160	93	50	82
	5	0.23	265	73	0	30
	total		2047	354	254	387
Black-tailed Godwit	1	0	0	0	0	0
	2	0.03	86	40	56	67
	3	0.08	732	216	222	308
	4	0.19	1652	136	73	120
	5	0.37	420	118	0	48
	total		2890	510	351	543

Northern Lapwing	1	0	0	0	0	0
	2	0.09	230	119	168	200
	3	0.24	2208	649	666	925
	4	0.34	3148	244	131	215
	5	0.67	765	213	0	88
	total		6351	1225	965	1428

5.5. Discussion and conclusion

One aim of this study is to pinpoint the potential impacts of land use changes to species habitats of agricultural landscapes. In contrast to other studies that often model species habitats based on past or present habitat conditions (e.g. SEOANE ET AL. 2004; THOMPSON ET AL. 2004; BAYLISS ET AL. 2005) this assessment considers potential future land use changes. This necessitates the use of special scenarios to spatially extrapolate the landscape changes, which methodologically extends already existing habitat suitability models.

Agricultural land on the Eiderstedt peninsula is traditionally dominated by extensively used grassland even though the share of grassland in relation to arable farmland was fairly variable in the past. The knowledge of past land use changes and its regional causes are important for the development of future scenarios. The scenario analysis applies three possible paths of land use development on Eiderstedt in the next couple of decades. In all of them the share of arable farm land ends up at two thirds of the entire agricultural land of the peninsula. Our assessment demonstrates the possible ecological impacts of such land use change. The results show that the pattern of agricultural land conversion has a great influence on the ornithological species composition in this area. It is our intention to raise the awareness about the potential implications to the environment that might be caused by political decisions. Therefore, the three scenarios purposefully represent very far-reaching developments. But even though the scenarios appear extreme, they are by no means unrealistic, as they are based on real statements by local interest groups that traditionally have a strong influence on decisions in regional politics in northern Germany.

The main difference between the three scenarios lies in the degree of fragmentation of the remaining grassland patches and their location. A conversion of agricultural land starting along existing roads leads to the highest degree of fragmentation, which potentially reduces the quality of the unconverted grassland as potential breeding habitat for birds. On the other hand, a conversion to arable farm land from east to west leaves intact larger areas of grassland in western Eiderstedt but the value of the bird sanctuary near Kotzenbüll is reduced due to its isolation and large shares of the remaining breeding habitats lie in the vicinity of a major tourist destination, which is likely to lead to considerable anthropogenic disturbances.

The potential environmental impacts of the land use conversion differ depending on the resulting distribution pattern of agricultural land. The ornithological impacts are quantified using the *HaSI* assessment scheme. Such *GIS*-based modelling techniques that rely on rule-based parameter combinations are considered to be effective tools in this context (POWELL ET AL. 2005; THOMPSON ET AL. 2004). The *HaSI* scheme is validated using bird abundance maps. The methodology of the *HaSI* assessment has a high accuracy because the *HaSI* values correlate well with the observed breeding pair density data of the selected bird species. In regions with a high *HaSI* the breeding bird density is also highest and a low *HaSI* corresponds to a low breeding pair number. Assuming a time independence of the species-specific breeding bird densities, the potential development in the number of breeding pairs supported by the habitats on Eiderstedt can be evaluated.

The potential decline in breeding pairs is particularly strong for Common Redshank, the species with the lowest abundance to start with (cf. Tab. 5.4). In all scenarios, its reduction is above average, making the species highly endangered of extinction in this region if breeding habitats were to be reduced as projected. The more abundant species appear to be slightly more resilient to the altered extent of suitable breeding habitats. Both the Eurasian Oystercatcher and the Northern Lapwing partly offset the reduced habitat availability by increasingly utilizing land area with only marginal suitability for breeding.

However, since the number of breeding pairs of all species assessed is reduced drastically in all three scenarios, it can be deduced that the overall quality of the

Eiderstedt peninsula as habitat for meadowbirds deteriorates considerably. The main reasons are the increasing isolation of suitable breeding areas and the increasing likelihood of disturbances by anthropogenic activities. The results indicate that not even the declaration of the three bird sanctuaries on Eiderstedt can offset this development since the suitability of these areas as breeding habitat is also critically impaired owing to the land conversions in the neighbourhood of these protected sites. Therefore, buffer zones around these bird conservation areas are of paramount importance to preserve the existing habitat quality.

The same holds for the salt marshes outside the main dikes, where the highest bird densities are generally observed. For these areas, suitable conditions for breeding need to be present in the adjacent hinterland as well if their overall quality as ornithological habitat is to be maintained. The importance of an intact neighbourhood is augmented if the impacts of sea level rise on the salt marshes are considered as well. Because of impossibility of retreat due to anthropogenic infrastructure such as dikes, an accentuated erosion of the salt marshes might take place, leading to the deterioration or complete loss of the potentially most valuable breeding areas for meadowbirds. The hinterland on the Eiderstedt peninsula could serve as highly suitable substitution habitat, but only if current conditions are preserved.

Based on the results of this assessment it is possible to identify the characteristics of an optimal bird conservation area on this peninsula, considering not only the habitat suitability for bird species, but also respecting the recent and future socio-economic developments of the local actors via participatory analyses. This study serves as a starting point for such assessments as it provides a model to analyze the potential impacts of land use changes. Moreover, the utilization of scenarios as presented here can help improve the efficiency of integrated land use planning and conservation management of landscapes. By considering potential landscape developments, such scenarios allow the formulation of optimal targets for a given region.

The scenario analysis illustrates that a much smaller number of breeding birds will be supported by the remaining suitable habitats if land use changes occur as projected. Today, many farmers argue that a distinct expansion of arable farming is the only way to survive economically and that shifts in the overall structure of

the regional agriculture necessitate these conversions. However, the Eiderstedt peninsula is not only an agricultural region but also a famous tourist destination because of its vast grassland areas and high densities of breeding or migrating birds. It is likely that large scale conversions of grassland to arable farm land also have an influence on the appearance of the Eiderstedt landscape to visitors. An assessment of the impacts of land use change on tourist activities on Eiderstedt is beyond the scope of this analysis and will be conducted in a separate study. Future assessments of land use change on Eiderstedt will further enhance the understanding of the impacts of planned land conversions, so that hopefully, farmers, birds, and tourists will all find or retain their optimal niches on Eiderstedt in the next decades without having to experience too extensive economic and ecological losses.

5.6. Appendix

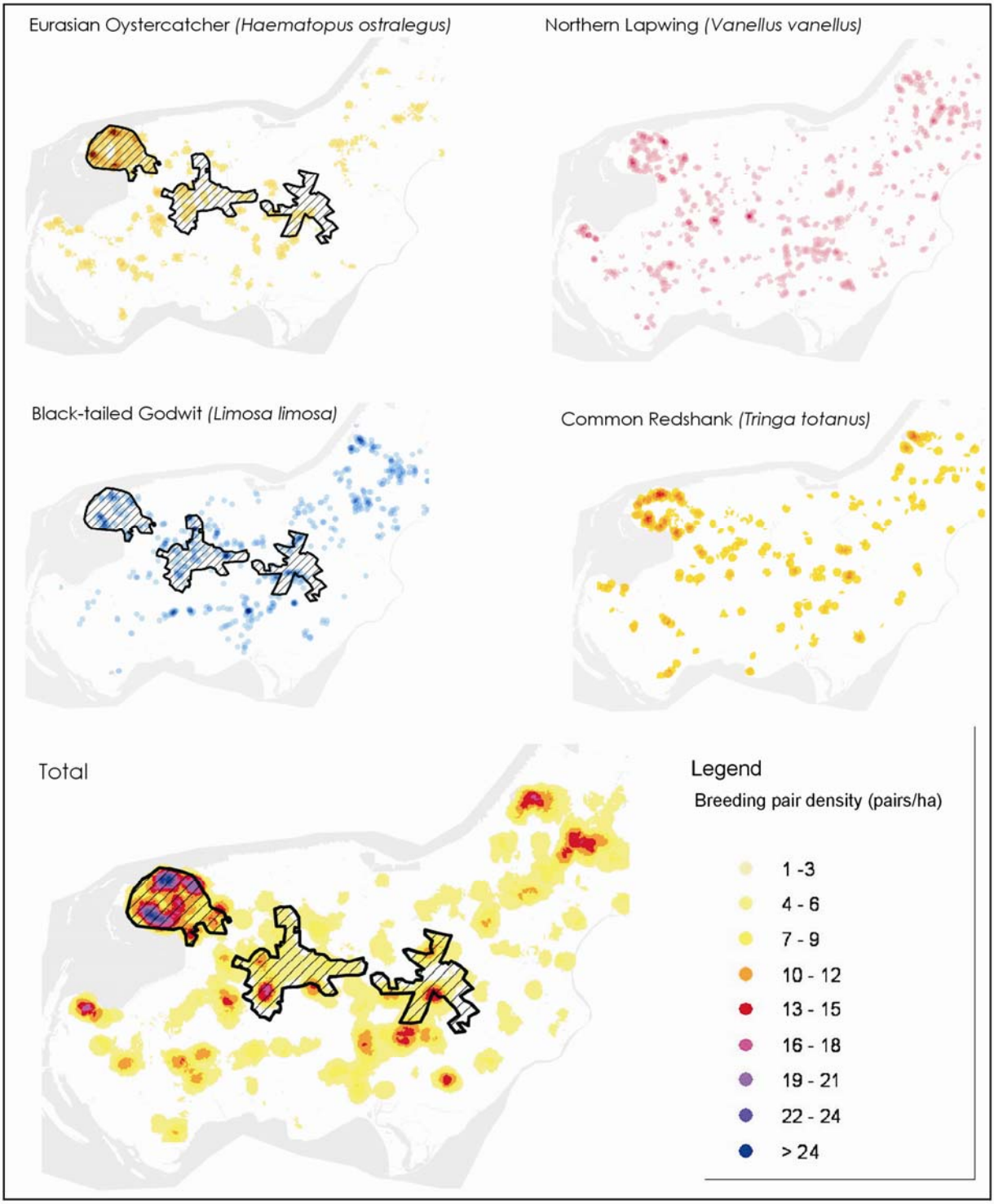


Fig 5.4 Overview of the spatial distribution of breeding pair densities for each considered bird species and in total of all four bird species. The density of breeding birds increases with darker and more saturated colours.

Part III

Integrated wetland distribution modelling for optimal land use options



Introduction

Changing land use for food and, increasingly, fuel production constitutes a great challenge to nature conservation. Conservationists are concerned that promotion of bioenergy plantations in the context of climate change mitigation policies could further threaten nature reserves and would lead to biotope loss. In the past, the integration of conservation concerns has often been ignored in agricultural as well as forestry production land use models. The purpose of the studies in the third part of this thesis is to evaluate both biophysical and economic potentials to preserve existing wetlands, to restore formerly native wetlands, and to create non-native managed inland wetlands in the EU-25 countries. Wetlands constitute valuable ecological resources. Globally, freshwater wetlands occupy less than 6% of the Earth's surface, but they provide 40% of the Earth's renewable services due to the interactions of physical, biological and chemical components (ZEDLER 2003). But, economically profitable land utilization requires drainage of the wetlands and therefore leads to intensive wetland degradation. Today, wetlands are considered to be among the world's most threatened ecosystems. Besides biotope loss and habitat fragmentation wetland loss also results in unprecedented flooding events and species declines (DAHL 2006).

Wetlands serve as buffer zones between terrestrial and aquatic ecosystems. Often this has been overlooked in land use planning and decision-making in the past (CALHOUN 2007), maybe due to the historical context, when wetlands were seen as wastelands and sources of disease (DAHL 2006). The recognition of the importance of the intersection of terrestrial and aquatic ecosystems in wetlands for landscape planning has emerged only recently (LINDENMAYER & HOBBS 2007). In Europe, the spatial distribution of wetlands is not well known except for large wetland areas or for wetlands of special ecological interest (MEROT ET AL. 2003). Even those wetland areas, which have been identified on the behalf of European Environment Agency (*EEA*), correspond to wetland areas of ecological interest and represent only a rather small part of all wetland areas (BERNARD 1994). The sixth chapter of this thesis deals with the development of the GIS-based wetland distribution model "SWEDI". By considering the matrix characteristics the model evaluates the spatially explicit distribution of existing wetland habitats and potential restoration sites. It simultaneously distinguishes different wetland types.

The existence of wetlands is not only driven by site specific natural conditions but also by the economic environment. Acquiring property for conservation can be expensive yet land costs have not received much consideration in designs aimed at expanding reserve networks (NEWBURN ET AL. 2005). Efficient reserve systems are also important due to limited conservation budgets. This study integrates both, biophysical and economic aspects, by linking the described GIS-based wetland model "SWEDI" to the European Forest and Agricultural Sector Optimization Model "EUFASOM" (SCHNEIDER ET AL. 2008). Nature conservation in the absence of humans will not succeed, all the more is of importance to establish conservation concerns also in the spatial scale of policy and economic market decisions (WIENS 2007). Chapter seven makes an important contribution to this topic.

The eighth chapter aims to identify the most suitable sites for reintroduction of wetlands by use of a GIS-based modelling approach that depends on a selection of restoration goals. There is an increasing interest in the reconstruction or restoration of wetlands in Europe. Since the members of the European Union decided to conduct an ecological network (*Natura 2000*) to fulfil the objectives of the *Habitats Directive* (92/43/EEC), this topic has become important in European conservation concerns. As existing reserves are inadequate to achieve conservation goals, it is necessary to acquire additional land with habitat value or restoration potential. Ecosystem restoration is also a vital tool in the maintenance and restoration of resilience. It could become particularly important as climate change progresses in the future and there are shifts in the ranges of biota (HARRIS ET AL. 2006). Often the methods and mechanisms by which restoration sites are identified have not been defined (BURNSIDE ET AL. 2002). In many cases the spatial attributes of the reserve system are not taken into account. Most models are based on the geographic distributions of species in relation to site cost or area, and as such they are likely to be collections of scattered sites. Yet, the spatial relationship among different components of a wetland system affects its persistence through its influence on species and vegetation dynamics. Spatial attributes that may affect these habitats include wetland size and shape, the number of reserves, and the distance between them, as well as wetland density and isolation (GIBBS 2000; WILLIAMS 2008). Therefore, integrated spatial habitat models are increasingly recommended as important elements of planning (BURGMAN ET AL. 2007). This study that is described in the third part supports strategic landscape evaluation and provides a method for identifying areas for targeted restoration.

6. Spatial distribution modelling of wetland potentials in Europe

6.1. Introduction

Socio-economic considerations and economic activities play an important part in land use management and conservation planning. But whereas land use action is often directed at broader scales, the focus in conservation management has been mainly on finer scales by neglecting a more holistic view of the landscape (FRANKLIN & SWANSON 2007). One reason is often a lack of accurate and consistent basis data. Therefore, conservation studies that match the scale of land use are often recommended, but rarely realized (SCOTT & TEAR 2007; WIENS 2007). There is also a growing demand of policy makers and researchers for high-accuracy landscape information at the European level (WASCHER 2000; KLIJN 2002). This study contributes to this problem by creating and preparing a wetland distribution model for integration into the bottom-up land use assessment EUFASOM, which is used to study synergies and trade-offs between wetland conservation efforts, greenhouse gas mitigation options including carbon sinks and bioenergy, and agriculture and forestry of Europe (SCHNEIDER ET AL. 2008). Through EUFASOM, economic wetland potentials for optimal land use options under certain policy scenarios are determined per country. Before this can be realized existing wetland areas as well as maximum area potentials for wetland restoration actions need to be evaluated in the European dimension. For this reason we intended to develop a methodology to locate existing wetland habitats as well as to model potential convertible sites for restoration of wetland biotopes.

In Europe, the spatial distribution of wetlands is not well known except for large wetland areas or for wetlands of special ecological interest (MEROT ET AL. 2003). The US CENTER FOR WATERSHED PROTECTION (2006) published an annotated bibliography of wetland research and another bibliography by WRIGHT (2007) gives an overview of GIS and wetland science. Most wetland studies deal with water quality and quantity assessments at local, regional or watershed scale (BRINSON & RHEINHARDT 1996; RUSSEL ET AL. 1997; BHADURI ET AL. 2000; BARTOLDUS 2000; RICHARDSON & NUNNERY 2001; WINTER & LABAUGH 2003; VEPRASKAS ET AL. 2005), others

concentrate on wetland functions and values (BRINSON 1993; BAYLESS 1999; MITSCH & GOSSELINK 2000; LEIBOWITZ 2003) or on vegetation and biodiversity patterns (MILLER & ZEDLER 2003; KIVIAT & MACDONALD 2004; MAHANEY ET AL. 2004). These studies are of paramount importance and significance for building reliable distribution models by improving model input data qualities.

Numerous studies about distributional modelling have arisen during the last decades, the majority of them specified on predicting habitat or species potential distributions (e.g. MCKAY 2001; LINDENMAYER ET AL. 2002; ANDERSON ET AL. 2003; CHEFAOUI ET AL. 2005; POWELL ET AL. 2005; WESTPHAL ET AL. 2007). GUISAN & ZIMMERMANN (2000) give an overview of these models and review their applications. Some of these models predict the spatial distribution of vegetation across a landscape based on its relation to relevant environmental variables (FRANKLIN 1995). Others simulate spatial distributions of single or multiple species to whole biomes or biodiversity in general (FISCHER 1990; PRENTICE ET AL. 1992; NEILSON 1995; FRASER 1998; MANEL ET AL. 1999). But only a few (e.g. TONER & KEDDY 1995; VAN HORSSEN ET AL. 1999; MITSCH & WANG 2000; MEROT ET AL. 2003; VAN LONKHUYZEN ET AL. 2004) concentrate on the distribution modelling of whole ecosystems, namely wetlands. For example, MEROT ET AL. (2003) tested a climato-topographic index for wetland distribution at catchment's level and OLSZEWSKA & TONDERSKI (2004) used a GIS to locate sites for wetland restoration in southern Sweden.

Some wetland studies modelled wetland distribution at global scales (MATTHEWS & FUNG 1987; ASELMANN & CRUTZEN 1989; STILLWELL-SOLLER ET AL. 1995; GLOBAL LAND COVER 2000; LEHNER & DÖLL 2004). Due to its global perspective, the spatial resolution is coarse and wetlands are seldom differentiated in detail. A critical overview of existing data on wetland distribution in Europe is illustrated in table 6.1, which shows similarities and differences between them. It becomes clear that no digital land cover or vegetation map of the EU exists that shows a detailed wetland distribution. Most studies concentrate on non-forested peatland only (e.g. MONTANARELLA ET AL. 2006). The most detailed information about wetland habitats in Europe offers the EUNIS (*European Nature Information System*) Database with the distinction of over 2600 terrestrial habitat classes at the fourth level (MOSS & DAVIES 2002 a,b). However, the corresponding EUNIS habitat type map (EUROPEAN TOPIC CENTRE ON BIOLOGICAL DIVERSITY (ED.)) that has been created using mainly aggregated CORINE data refers

only to the first level (= 10 major habitats) of the EUNIS habitat classification. Also, the CORINE biotopes data (EUROPEAN COMMISSION 1991; MOSS & WYATT 1994, MOSS ET AL. 1996; EEA 2000a) that are based on reported NATURA2000 sites do not represent the existing wetlands completely and are only available in terms of spots on the map without area size statements. At present, the CORINE data (EEA 2000b) is the most detailed land cover database covering the European Union. One disadvantage is the heterogeneity of the classes determined by functional land use and not by land cover itself. The digital map of the potential natural vegetation of Europe (BOHN & NEUHÄUSEL 2003) shows the most detailed classification and potential distribution of wetland vegetation types across Europe but irrespective of human influences. River regulations, peat extraction or urbanisation on former wetland areas often lead to changed potentials in wetland restoration.

Other European wetland data exist on country or regional basis as statistics. But aggregating statistical and spatial data from many sources into one database, as is sometimes done in broad scale studies often causes low spatial accuracy, because the sources differ in spatial accuracy, reliability, acquisition data and class definition.

The aim of this study is to compile spatially consistent information on wetlands differentiated by wetland types and characteristics, but initially regardless of their conservation status or restoration costs. We distinguish between existing wetland habitats and potential convertible sites for restoration of wetland biotopes by considering actual land use options. This model provides an important prerequisite for the further development of a cost-effective wetland site-selection model of an EU-wide ecological network of wetland sites. The basis of this model is the optimal combination of existing spatial datasets to obtain the spatial distribution of wetlands by definition of flexible knowledge rules (cf. MÜCHER ET AL. 2004).

Table 6.1 State of the art in European wetland mapping – an overview of other wetland geodata

Wetland data	Resolution	Wetland types	EX	POT	Region	Comments
CORINE Land Cover 2000	100 m	- moors and - heathland - inland marshes - peat bogs - open waters	X		EU	- no wetforests - category “moors and heathland” is not necessarily wetland
CORINE Biotopes	Point data	differentiated classification	X		PHARE countries	- only selected (protected) wetlands - EU-Region and not consistent - replaced by EUNIS
European Nature Information System (EUNIS)	1 km / Point data	- inland surface waters - mires, bogs, fens	X		Europe	- numerous subtypes but not spatially explicit - point data: CORINE Biotopes; Natura 2000, not consistent - map base: extended CORINE LC - no wetforests
EURASIA land cover characteristics data base	1 km	- differentiated wetlands	X		Eurasia	- no wetforests - broader wetland areas only
Baltic GIS	(1km) 50 km	- undifferentiated wetlands	X		Baltic Sea drainage basin	- regional wetland area statistics with wetland locations represented in DCW (Digital Chart of the World) land cover layer = average wetland area/50 km²
PELCOM	1 km	- undifferentiated wetlands - water bodies	X		Europe	- no wetforests
Map on Potential Natural Vegetation of Europe	1 km	differentiated by vegetation type		X	Europe	- without human influence
PEENHAB	10 km	- freshwater Habitats - raised bogs, mires, fens		X	Europe	- subtypes not applied for wetlands - uses SynBioSys for habitat prediction. - low spatial resolution

6.2. Methods

6.2.1. Definition of wetlands

Often wetland terms and definitions are not standardized. The RAMSAR Convention (Article 1.1) defines wetlands as "areas of marsh, fen, peat land or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six meters". In addition, the Convention (Article 2.1) determines that wetlands "may incorporate riparian and coastal zones adjacent to the wetlands". In wetlands water is present at or near the surface of the land also if only for varying periods of the year. Wetlands vary widely in soil, topography, climate, hydrology, water chemistry, vegetation, and other factors, also because of human disturbance (POTT & REMY 2000; DIERSSEN & DIERSSEN 2001; BLUME ET AL. 2002). In our study we concentrate on the natural freshwater or inland wetlands as defined in table 6.2.

Table 6.2 Used wetland terms and their definitions, based on COWARDIN ET AL.(1979), MITSCH (1994), SANDERSON (2001).

Common Wetland Names	Definition
Peatland	generic term of any wetland that accumulates partially decayed plant matter.
Bog	peat-accumulating wetland that has no significant inflows or outflows. Water and nutrient input entirely through precipitation; characterized by acid water, low alkalinity, and low nutrients. Peat accumulation usually dominated by acidophilic mosses, particularly sphagnum.
Fen	peat-accumulating wetland that receives some drainage from surrounding mineral soil. Usually dominated by sedge, reed (→reedswamp), shrub or forest (→swampforest). Surface runoff and/or ground water have neutral pH and moderate to high nutrients.
Marsh/ natural wet grasslands	permanently or periodically inundated site characterized by nutrient-rich water and emergent herbaceous vegetation (grasses, sedges, reed) adapted to saturated soil conditions. In European terminology a marsh has a mineral soil substrate and does not accumulate peat.
Reedswamp	marsh or fen dominated by *Phragmites* (common reed);
Swampforest	wetland dominated by trees, most often forested fen. Depends on nutrient-rich ground water derived from mineral soils.
Alluvial forest	Periodically inundated forest areas next to river courses.

The definition of inland wetlands also includes marshes and wet meadows dominated by herbaceous plants that are most often human made as well as shrub- or tree-dominated swamps. In the following text we will refer to them as wetlands only. In Europe, inland wetlands are most common on floodplains along rivers and streams, along the margins of lakes and ponds, and in other low-lying areas where the groundwater intercepts the soil surface or where precipitation sufficiently saturates the soil (vernal pools and bogs). Many of these wetlands are seasonal and may be wet only periodically. The quantity of water present and the timing of its presence partly determine the functions of a wetland and its role in the environment (MULAMOOTTIL ET AL. 1996).

6.2.2. The spatial wetland distribution model "SWEDI"

GIS and spatial modelling are assumed to provide an appropriate tool to locate existing wetland areas as well as to identify the most suitable areas for wetland regeneration measures. This GIS model aims to depict the distribution of wetland areas at regional level and at coarse geographic scale. This involves the integration of a variety of GIS datasets and multiple iterations of expert review and interpretation to delineate the potential wetland areas of Europe. We used the GIS tool ArcGIS9 for analysis. Figure 6.1 gives an overview of the SWEDI (*Spatial wetland distribution*) model structure and its core input data.

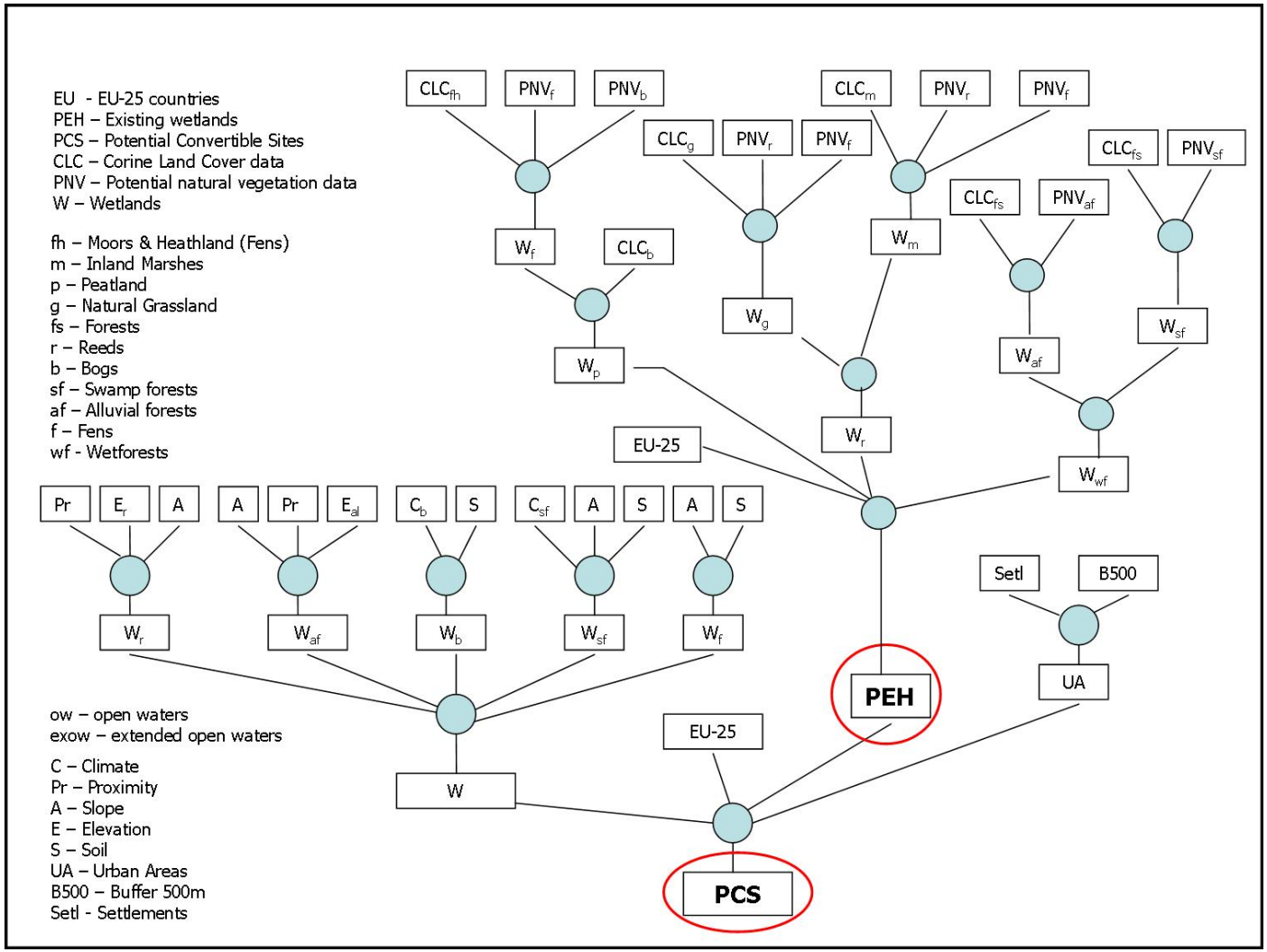


Fig. 6.1 The spatial wetland distribution model “SWEDI”.

The following methodological description of the model is subdivided into two parts. The first deals with the spatial distribution evaluation of existing wetlands in Europe, and the second with the modelling of potential convertible sites for wetland restoration management.

Existing wetland habitats (PEH)

Existing wetland biotopes are defined as areas where wetlands with state close to nature actually appear within Europe. The following equations schematically represent the theoretical methodology for the evaluation of existing wetlands in the EU-25. The analysis is executed with Model Builder and the Spatial Analyst Extension of ArcGIS9.

$$PEH = (Wp \cup Wr \cup Wwf) \cap EU\text{-}25$$
$$Wp = Wf \cup CLCb$$
$$Wf = (CLCfh \in PNVf) \cup (CLCfh \in PNVb)$$
$$Wr = Wg \cup Wm$$
$$Wg = (CLCg \in PNVr) \cup (CLCg \in PNVf)$$
$$Wm = (CLCm \in PNVr) \cup (CLCm \in PNVf)$$
$$Wwf = Waf \cup Wsf$$
$$Waf = CLCfs \cup PNVaf$$
$$Wsf = CLCfs \cup PNVsf$$

Abbreviations:

CLC – Corine Land Cover data

EU – EU-25 countries

PEH – existing wetlands

PNV – potential natural vegetation data

W – wetland

b – bogs

f - fens

fh – moors & heathland (Fens)

g – natural grassland

m – inland marshes

p – peatland

r – reeds

sf – swamp forests

af – alluvial forests

wf – forests

The Corine land cover map 2000 with spatial resolution of 100 m serves as core base map (BOSSARD ET AL. 2000; EEA 2000b). From the CORINE data, the following land cover classes have been extracted: moors & heathland (3.2.2.), inland marshes (4.1.1.), peat bogs (4.1.2.), inland waters and estuaries (5.1. and 5.2.2.), natural grassland (3.2.1.) and forests (3.1.). The Commission of European

Communities (1995) gives detailed definitions of each class. Inland waters and estuaries were selected for illustration purposes and future evaluation of biotope complexes. Within the spatial model the land cover class “peat bogs” serves as the only one that does not need to be altered to show existing “bog” wetlands, whereas all other selected land cover classes have to be split up separately. Out of all European forests, for example, we wanted to select only the wetforests, namely alluvial forests next to river courses and fen or swamp forests. In addition, moors, wet heaths and riverine and fen scrubs have to be extracted from “moors and heathland”, and “natural grassland” as well as “inland marshes” serve as base data for the model parameter “natural wet grasslands”. For example, the EUNIS habitat classifications consider over 150 classes of the CORINE class “natural grasslands”, including wet grasslands (E3) and wet tall-herb and fern stands (E5.4. and E5.5.). We aimed to separate the wet grasslands from the natural grasslands. For this reason, the map of the potential natural vegetation (PNV) of Europe (BOHN & NEUHÄUSEL 2003) has been selected as source to locate those sites covered with existing wetlands. The PNV map in general distinguishes following wetland types: a. tall reed vegetation and tall sedge swamps, aquatic vegetation (PNV class R), b. mires (S), c. swamp and fen forests (T), d. vegetation of flood plains, estuaries and fresh-water polders and other moist or wet sites (U). These types can then be further subdivided. We extracted these wetland types and intersected them with the CORINE data. Only those sites matching both attributes were considered as present existing wetland site. The remaining sites were assumed to be non-wetland. However, this does not exclude the probability of the non-wetland areas to be potential wetland restoration sites as is explained in more detail below. The SWEDI model results of existing wetlands are illustrated through wetland distribution maps (see figure 6.3 and 6.5.).

Accuracy assessment of existing wetlands

In order to verify the validity of the distribution of existing wetlands in SWEDI, model predictions must be compared with an independent data set (VERBYLA & LITAITIS 1989; ARAUJO ET AL. 2005). Some validation methods for habitat models are discussed in FIELDING & BELL (1997), MANEL ET AL. (2001), BOYCE ET AL. (2002), or OLIVIER & WOTHERSPOON (2008) as well as in OTTAVIANI ET AL. (2004). In this study, validation of the defined decision rules and resulting wetland

distribution maps is based on the use of the CORINE biotopes database and parts of the RAMSAR list of wetlands of international importance (2008) by using linear regression analysis. The CORINE biotopes (Version 2000) database is an inventory of major nature sites. The aim of the database was to enhance reliable and accessible information about vulnerable ecosystems, habitats and species of importance as background information for community environmental assessment (EEA 1995). The wetland sites of the database are among others attributed with the size of the wetland. Site coordinates are included for easy localization of the biotopes within a GIS. We selected 50 freshwater wetlands from the database and compared their occurrence in the SWEDI model considering spatial accuracy and wetland size. The same procedure has been applied to 50 selected RAMSAR sites.

It appears that the model of existing wetlands is reasonably accurate with regards to the samples of the independent data sets. All selected wetlands of the databases are also represented in SWEDI. The differences lie in the area extent of the respective wetlands: In over 70% of the cases the model overestimates the size of an existing wetland. One reason is the fact that the existing wetlands module of the SWEDI model accepts uncertainties about the state of the wetland ecosystem also due to scale reasons. We are not able to distinguish between afforestations or natural alluvial forests in a floodplain, for example, what also might lead to overestimation errors of the results. 26% of the sites are underestimated in size. Again the difficulty is the accurate demarcation of wetlands and its types from open waters and terrestrial land due to their dynamic characteristics and their fluctuating and undefined borders. Often open waters are integrated into the wetland definitions of the databases whereas these wetland types are considered separately in the SWEDI model. Astonishingly, more than 85% of the selected wetlands stay within the defined uncertainty range of 15% deviation.

Potential convertible sites (PCS)

The second part of the GIS assessment evaluates potential convertible wetland sites. These areas may be used for location of restoration programs or habitat creational measures. The distribution of wetlands is explained by many dependent and explanatory variables. Important factors are the climatic, hydrological, geological, environmental and socio-economic conditions of the area. The classification of wetland distribution is therefore preferably based on multivariate analysis of independent variables (GUISAN & ZIMMERMANN 2000).

The connection between the respective information of the database and the probable appearance of the wetlands is determined by assuming that there is a relationship between environmental gradients and wetland distribution (FRANKLIN 1995). We use traditional statistical methods based on observed correlation to analyse environment-wetland relationships as well as geographically weighted regression analysis. It is useful for European scale analyses, because it allows for regional differences in relationships by estimating regression parameters that vary across space (cf. WALTER & WALTER 1953; MILLER ET AL. 2007). Characteristic soil parameters, climate conditions, slope angles, and elevations are worked out for every wetland type on the basis of several literature resources (BRINSON 1993; KUNTZE ET AL. 1994; ELLENBERG 1996; SUCCOW & JOOSTEN 2001; BFN 2004). Through this, rule-based statements are derived about the potential appearance of the target wetland types. In combination with geographical data these statements allow the identification and localisation of potential wetland sites within a GIS.

In general, another important factor determining wetlands is the positive water balance of the site. The inflow of water to that area must be greater than the amount of water leaving the wetland by infiltration, outflow, or evaporation. Accurate information about that topic is not available at European scale. Additionally was assumed that hydrology is up to a certain extend anthropogenic convertible and manageable. Therefore, the water factor is only indirectly integrated into the model through climate and soil data. The following equations give an overview of the model and table 6.3 illustrates the factors that characterize each wetland type.

$$PCS = ((W \notin PEH) \cap EU\text{-}25) \notin UA$$
$$W = Wr \cup Waf \cup Wb \cup Wsf \cup Wf$$
$$UA = Set \cup B500$$

Abbreviations:

PCS – potential convertible sites

W – wetland

PEH – existing wetland

EU-25 – EU-25 states (excl. Cyprus and Malta)

UA – urban areas/human settlement areas

Set – human settlements

B500 – buffer of 500 m around cities

af – alluvial forest

b – bogs

f – fens

r – reeds

sf – swamp forest

Table 6.3 Rating factors that characterize each wetland type

	Soil	Slope Angle	Climate	Proximity to open waters	Elevation
Fen	X	X			
Bog	X		X		
Swamp Forest	X	X	X		
Alluvial Forest		X		X	X
Reeds		X		X	X

For the assessment an automated target area search and representation is developed for the potential convertible wetland areas using the Model Builder in ArcGIS9 and Idrisi. Former wetland areas are considered as most suitable for wetland recreation (ELLENBERG ET AL. 1991; WHEELER ET AL. 1995; SCHULTLINK & VAN VLIET 1997). These might be arable fields, pasture lands, fallow or forested areas on sites of former wetlands that have been intensely changed. Actual soil conditions might give hints for potential wetland biotopes. We use the European soil database (JOINT RESEARCH CENTRE 2004b) of 1 km grid resolution and extract following potential wet- and peatsoil-classes: gleysols, fluvisols, gleyic luvisols, histosols, gleyic podzol. In the model the wetland types bogs, swampforests and fens are considered to be soil dependent (see table 6.3).

Table 6.4 Wetland type characteristics concerning their climate ranges of occurrence.

Wetland type	Average annual temperature (in °C)	Average precipitation (mm/a)	Max temp av. warmest month (°C)	Min temp av. coldest month (°C)
Bogs	3 - 6	300 – 1 000	12 - 17	-15 – (-2)
	9 - 11	1 200 – 2 000	13 - 15	5 – 7
	3 - 8	1 400 – 2 400	10 - 12	-2 - 0
	5 – 9,5	550 – 1 500	14 - 19	-3 - 5
	4 – 5,5	900 – 1 400	11 - 12	-3 – 0
	3.5 – 7.8	530 - 630	17.5 - 19	-10 – (-2)
	- 10 - 1	200 - 500	8 - 13	-25 – (-10)
Aapa mires (fens)	- 3 - 5	250 - 700	8 - 15	-17 – (-5)
Transitional mires (fens)	0 - 5	500 - 870	8 - 14	-12 – (-3)
Degraded bogs, now wetforests	8 - 9	600 – 1 200	15 - 16	0 – 4
wetforests	6 - 11	450 – 1 000	16 - 21	-5 – 0
	14 - 15	> 1 000	20 - 22	6 – 8
	9 - 10	550 – 1 000	15 - 16	4 - 5

The climate parameter is only applied for the parameters bogs and swampforests; all other wetland types are rated as azonal and therefore relatively climate independent (SUCCOW & JESCHKE 1990; ELLENBERG 1996; WALTER & BRECKLE 1999). The climate variables of the wetland types shown in table 6.4 are extracted from the explanatory text of the map of the Natural Vegetation of Europe (BFN 2004) and are mainly based on WALTER & LIETH (1967). We use the attributes temperature (max temp of warmest month, min temp of coldest month, average annual) and precipitation (average annual) of the Bioclim and Worldclim data at spatial grid resolution of 30 arc-seconds (~ 1 km²).

The analyses of elevation dependent wetland types might also refer to climate conditions (MEROT ET AL. 2003). However, we are confined to the statements of highest occurrences of respective wetland types by the explanatory text of the PNV map of Europe (BFN 2004). The base elevation data for Europe are taken from GTOPO30 data a global digital elevation model at spatial resolution of 30 arc-seconds (sheets: W020N90, E020N90, W020N40, E020N40) (USGS 1996). In addition to that the bio-geographical regions map of Europe (EEA 2002) contribute to the elevation parameter by dividing the height variables into several

bioclimatic regions that better reflect the height-limits than country based distinctions of regions. The map of bio-geographical regions is based on the PNV map (BOHN & NEUHÄUSEL 2003). It distinguishes between six bio-geographical regions in the EU-25, namely Alpine, Boreal, Continental, Atlantic, Mediterranean, and Pannonian. Alluvial forests and reeds are considered elevation dependent, less because of climatic conditions, but more due to loss of suitable ground conditions (ELLENBERG 1996; MULAMOOTTIL ET AL. 1996; BFN 2004). The climate dependent wetlands are assumed to limit their height occurrence by this parameter itself. An elevation constraint is therefore not necessary. Only the fens are assumed neither climate nor elevation dependent. They solely refer to soil conditions and the slope parameter. Table 6.5 shows the wetland type characteristics concerning their maximum elevation occurrence range.

Table 6.5 Wetland type characteristics concerning their maximum elevation occurrence range.

Wetland type	Biogeographical Region	Elevation (m)
Reeds	Boreal, alpine (scand.)	<= 500
Reeds	Alpine (other), all others	<= 800
Alluvial forests	Boreal, alpine (scand.)	<= 500
Alluvial forests	Alpine (other), all others	<= 1 200

The slope parameter is evaluated based on the elevation data using the Spatial Analyst extension of Arc GIS9. Only those areas with a slope angle below 1° are assumed suitable for the wetland types reeds, alluvial forests, swampforests and fens (MULAMOOTTIL ET AL. 1996; LYON 2001). Due to scale reasons the slope angle is set to this maximum extension and does not distinguish slope angles below that point as has been done in case studies of larger scale (TSIHRINTZIS ET AL. 1998; HELMSCHROT & FLÜGEL 2002).

Also the proximity to inland waters or to existing inland peatland is an important criterion for localisation of target areas if other parameters are fulfilled. We establish multiple ring buffers around inland waters and other bog areas of 500 m radius. The extension of potential water surrounding wetland sites like alluvial forests can be detected by a combination of the proximity with other parameters.

Highly populated areas as are towns and cities provide very limited space for wetland restoration or construction. For this reason, potential convertible sites are

only modelled for agriculturally used areas, grasslands and forests by using pseudo-absences for urban areas (cf. CHEFAOUI & LOBO 2008). Urban areas including a buffer zone of 800 metres are omitted by the model. We use the CORINE Land Cover 2000 data for determination of these sites.

Finally, the existing wetland sites are subtracted from the preliminary results to obtain only data on potential convertible areas. All data encompass the whole EU-25 states boundaries with exception of Malta and Cyprus that are not included in the analysis.

6.3. Results

The results of the SWEDI model are illustrated through wetland distribution maps. It is possible to produce the existing habitats at spatial resolution of one hectare and the potential convertible sites at 1 km² grid for the EU-25 states excluding the islands Malta and Cyprus. Figure 6.2 shows the spatial distribution of existing habitats (dark grey) and potential convertible sites (light grey). Exemplarily, some details are extracted to illustrate the high spatial resolution of the model.

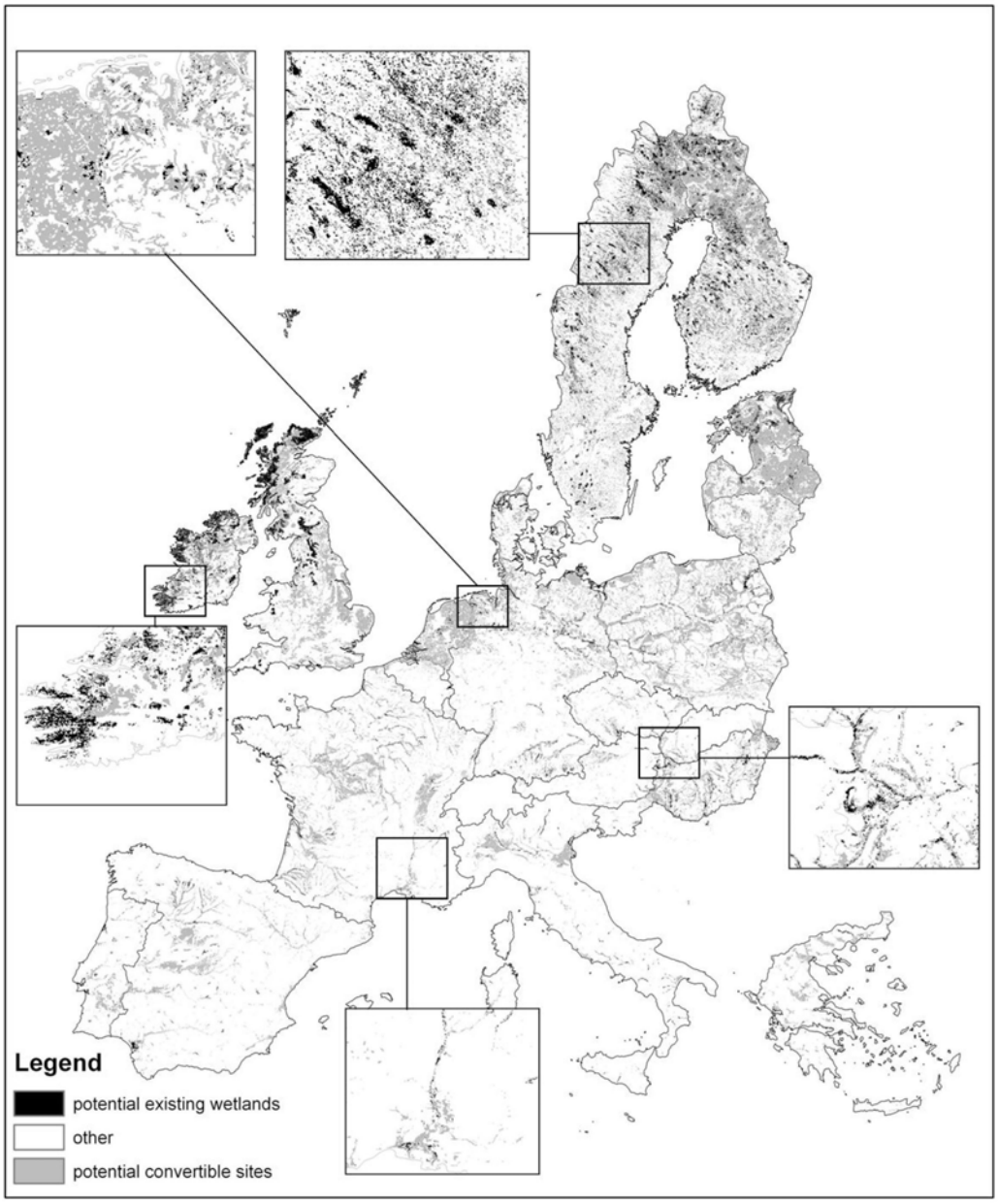


Fig. 6.2 Map of the spatial distribution of existing habitats (dark grey) and potential convertible sites (light grey) with more detailed examples.

The map of figure 6.2 reveals that the majority of existing wetland areas (PEH) is situated in the northern and western European countries, while the potential convertible sites (PCS) are well distributed over the EU. In total, about 4% of the EU-25 land area consists of potentially existing wetlands and an additional 21% of the land areas are potential convertible to wetland sites. This constitutes a maximum share of wetlands of one fourth of the total land area of the EU-25. Figure 6.3 gives an overview of the total area (in 1 000 ha) of existing and the potential convertible wetland sites per country.

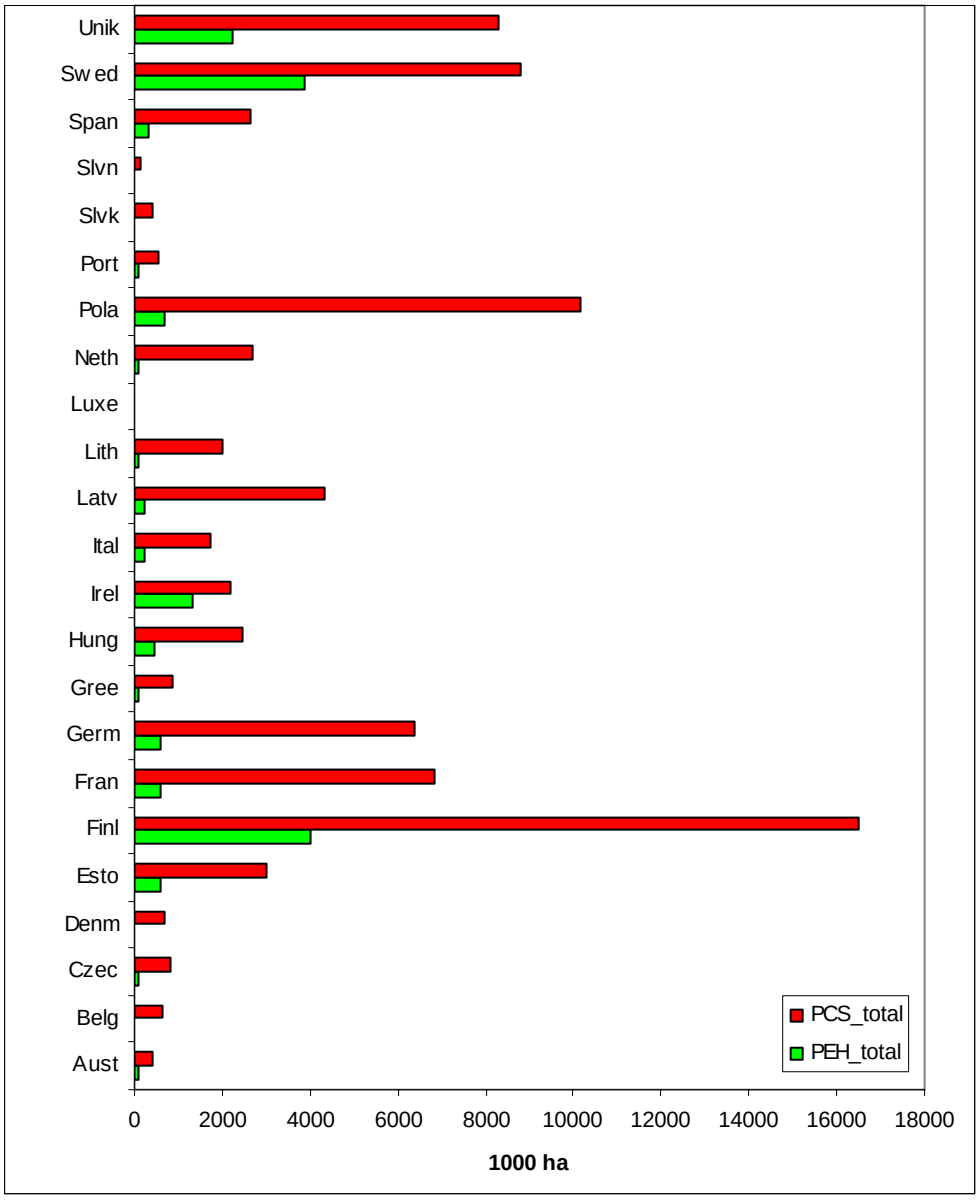


Fig. 6.3 Total wetland area (in 1000 ha) per country.

Open waters are excluded from the evaluation. Finland and Sweden own by far the most extending existing wetland areas with about 3.8 million ha wetlands. Also Ireland has great amounts of existing wetland areas (about 1.3 million hectares) but less in comparison to the Scandinavian countries. Finland and Sweden also lead in the amount of potential convertible wetland sites. In this category Poland, Great Britain as well as France and to a certain extent Germany as well show high amounts of land suitable for wetland restoration.

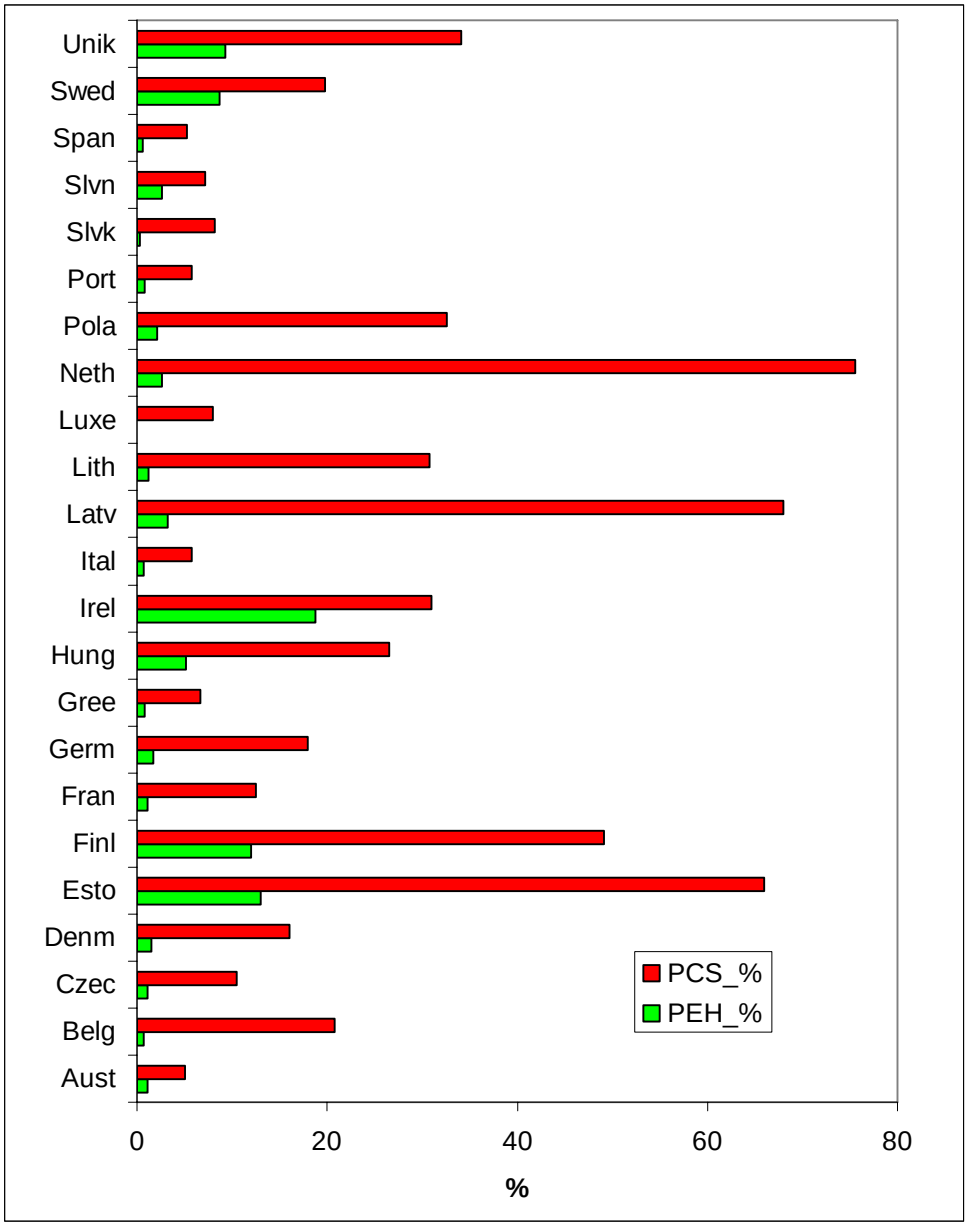


Fig. 6.4 Relation between country size and wetland area (%).

If we now look at the relationship between wetland areas and country size (see Figure 6.4) we get a different picture: Now Ireland shows the highest wetland rate (PEH) with about 19% of its country area, followed by Estonia (13%) and Finland (12%). Also Sweden and the UK with 8.7% and 9.25% of their country size own a high existing wetland rates in comparison to other countries whose amounts lie between 0.03% (Luxembourg), 0.6% (Spain), and 5.1% (Hungary), or 3.2% PEH of the country area in the case of Latvia, for example.

Concerning the PCS per country area, Latvia (68%), the Netherlands (75.6%), and Estonia (66%) have the highest relative potentials. The PCS rate of Finland, Poland, Great Britain, and Ireland amounts to between 31 and 49% per country area. In this case Denmark, Sweden and Germany have potentials of about 16 to 20% and the PCS rate of all other countries amount between 5.1% as lowest rate in Austria and 12.5% in France.

In Annex 6.6., more detailed and coloured maps of the spatial wetland distribution model are attached. Here, we separate the main wetland types into three maps showing the spatial potentials of peatland (fen/bog), wetforests (alluvial forest/ swamp forest) and wet grassland. It becomes evident that the potential wetland restoration sites are often overlapping. This is especially true for peatlands. Moreover, some wetland types might be temporary successional vegetation states of others within the wetland biotope complexes. Main areas of existing peatland are the Scandinavian countries and Ireland. Here, as well as in Scotland, Eastern Poland and Estland also highest amounts of potential bog areas are found. All other illustrated potential peatland areas may be favourable for fen restoration. Fens can also be created on potential bog areas, but this constraint does not work vice versa. It is remarkable that the formerly extending bog areas of North-Western Germany, that have been mainly drained and exploited during the last centuries, show fen instead of expected bog potentials in SWEDI. This might be due to model uncertainties or errors, but can as well be a hint that the bogs have developed under different climatic conditions from the end of the last ice age and would now only be relicts. The destruction of these bog areas possibly means an unrecoverable demise of the ecosystem. Like wet grasslands also wetforests are found along water courses and in the proximity of other open waters. Especially the swampforests are constricted to wet soils and to specific climate conditions.

Main areas of potential swampforest sites are therefore found in Central and Eastern Europe but also in the UK. In northern and western European countries the wetforested area does not exceed the peatland areas whereas in Germany and Poland and further south wetforests are the most extending wetland types. Extending areas of potential wet grassland sites are shown in Scandinavia, Estland, and Ireland, but also in Hungary. Wetland areas need to have at least a size of one hectare to be included into the spatial model. Therefore, often reeds along lakeshores are not shown in the results and Finland even counts no wet grasslands even though there are reeds growing along some waters.

Summarizing the total PCS areas distinguished after the main wetland types for the European countries one gets following results: About 1 329 218 km² additional bog areas, 643 331 km² additional fen areas, as well as 305 671 km² additional wetforest areas can be created potentially.

6.4. Discussion

Over the last century, the number and size of wetlands in Europe has decreased dramatically (SCHULTLINK & VAN VLIET 1997). Mainly agricultural and forestry practices have caused the loss of wetland area in many European countries. Even though fens and floodplain forests have been opened up for cultivation since the early middle Ages, their major decrease has occurred during the last few decades of the last century and is still continuing (WHEELER ET AL. 1995). Today, most of the EU-15 wetlands are drained, degraded and cultivated (JOOSTEN & CLARKE 2002). It is estimated that only 30-40% of all wetlands existing at the beginning of the 20th century in Europe have remained (WHEELER ET AL. 1995; SCHULTLINK & VAN VLIET 1997). During the last decades an increasing interest has evolved in the restoration of former wetland areas what is confirmed through several schemes (*Birds Directive, Habitats Directive, and Water Framework Directive*). There are lots of reasons to promote protection and restoration of wetlands. Wetlands are among the world's most productive environments. They are important habitats for a wide range of wild plant and animal species depending on the wetland's productivity (BAUER 1997; RAMSAR BUREAU). In addition, wetlands perform many functions due to the interactions of their physical, biological and chemical components (SUCCOW & JESCHKE 1990; MITSCH 1994; VERHOEVEN ET

AL. 2006). These are for example storm protection and flood mitigation, as well as shoreline stabilization and erosion control. Wetlands influence also the water balance through water storage and groundwater recharge; and they further improve the water quality through retention of nutrients, sediments, and pollutants.

Despite numerous data on land use in Europe, a detailed analysis of the distribution of wetlands and potential restoration sites has been lacking so far. There is a growing demand of policy makers and researchers for high-accuracy landscape information at the European level. We developed a detailed wetland distribution map in European scale with high spatial resolution. Not only does it distinguish between different wetland types but also between existing and potential convertible wetland sites, which have not been available before. Whereas the evaluation of existing wetlands relies on a cross-compilation of existing spatial datasets, the potential wetland restoration sites are determined by definition of flexible knowledge rules in combination with geographical data. The orientation towards physical parameters and the allowance of overlapping wetland types characterizes the SWEDI model. The detailed spatially explicit wetland classification of the SWEDI model allows connections to other habitat databases, for example EUNIS, as well.

The accuracy of the SWEDI model is strongly restricted by the availability and quality of geographical data. For example, the soil information is generally poor and often misleading from the standpoint of wetland functionality. Another uncertainty is the state of the ecosystem of the PEH. In SWEDI we are not able to make statements about the naturalness of the site. This is the reason that we describe the existing wetlands also potential. Nevertheless, the validation with independent datasets of wetland biotopes proved high accuracy of the existing wetland sites in the SWEDI model and the area sizes are mainly reproduced within the uncertainty range. The utilization of GIS makes the methodology highly applicable and easily to improve concerning data sources.

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 an empirical distribution model to wetland ecosystems in European scale. These data

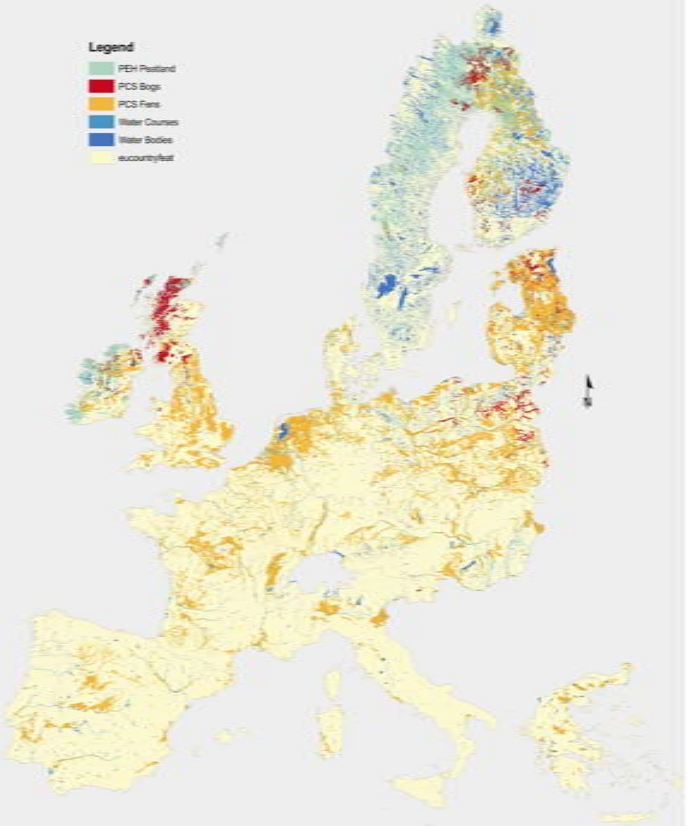
can be used as ground information for further studies, for example helping to locate sites suitable for renaturation programs, or for the introduction of faunistic corridors respecting the Natura 2000 network of sites. The application of the model in nature conservation issues favours the success in regional conservation planning. The SWEDI model on the other hand is meant to be integrated into the economic optimization EUFASOM model to evaluate the economic wetland potentials per EU-country (SCHNEIDER ET AL. 2008); furthermore it is going to be the base for biodiversity studies of endangered wetland species and is used as basis for a cost-efficient spatial wetland site-selection model.

Besides their physical and ecological values (MITSCH & GOSSELINK 2000), wetlands are part of the cultural heritage of people. They are important for recreation and tourism opportunities and enrich the landscape (NOVITZKI ET AL. 1997). Nevertheless, all these functions, values and attributes are only maintained if the ecological processes of wetlands are allowed to continue functioning.

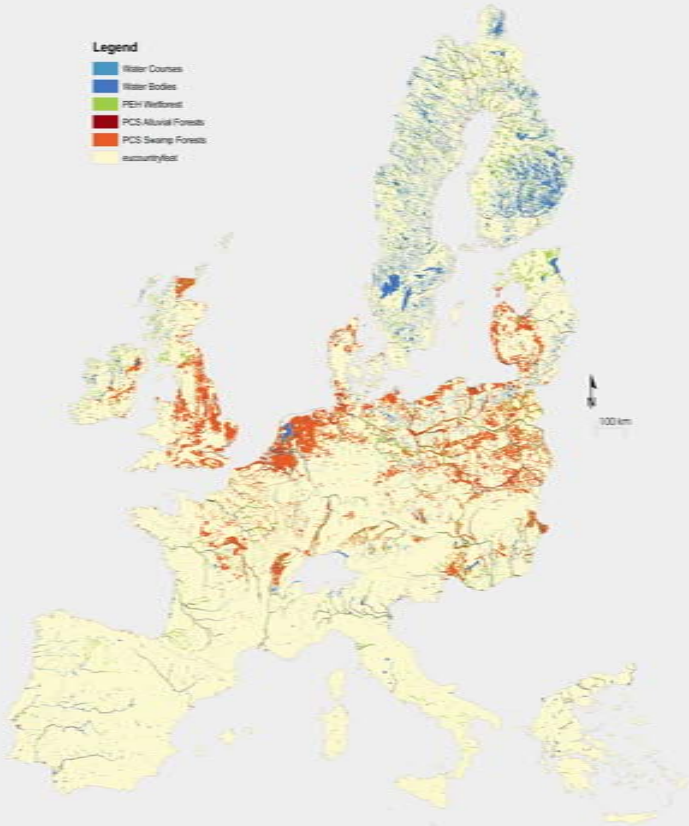
6.5. Appendix

Fig 6.5 Potential wetland distribution per wetland type

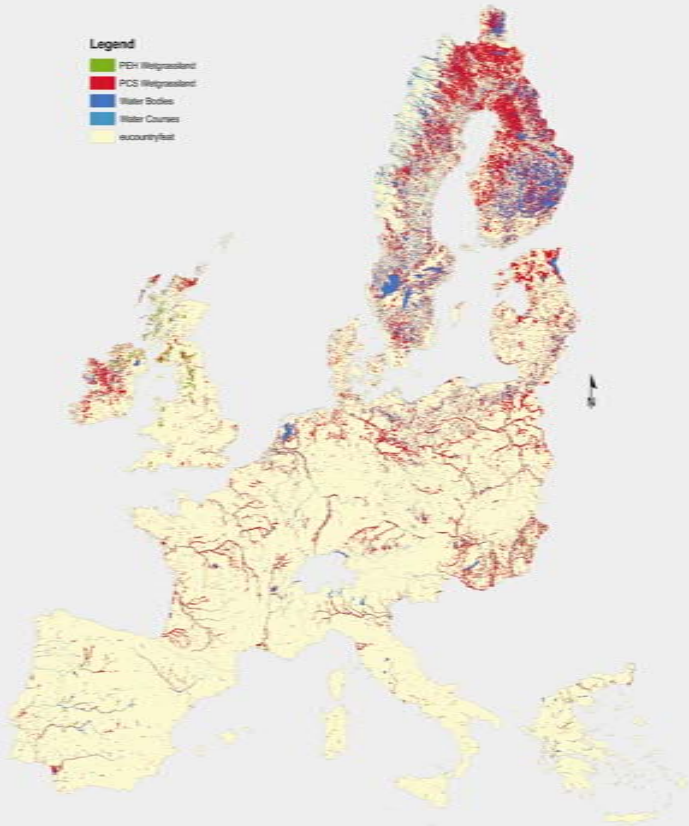
a. Peatlands and potential restoration sites



b. Wetforests and potential restoration sites



c. Wetgrasslands and potential restoration sites



7. Evaluation of European wetland restoration potentials by considering economic costs under different policy options

7.1. Introduction

This study focuses on inland freshwater wetlands of Europe. While fens and floodplain forests have been drained and cleared since the early Middle Ages, the main decrease in wetlands happened over the last century and is still continuing (RAMSAR COMMISSION). Ongoing drainage, conversion, pollution, and over-exploitation of the wetland resources make them to be among the world's most threatened ecosystems (JOOSTEN & CLARKE 2002). The last decades have seen increasing interest not only in wetland conservation but also in the restoration of wetlands. Restoration and conservation management are increasingly viewed as complementary activities and restoration measures are therefore often included in conservation management (YOUNG 2000; HOBBS 2005; MANNING 2007). Many existing reserves in highly modified human cultural landscapes are too small or too isolated to provide for the full biodiversity benefits. It is therefore necessary to acquire additional land with habitat value or restoration potential (MILLER 2007). Europe is densely populated in some parts and without protection and management, agricultural and forest demands would leave space for nature conservation in marginal areas only. Counteracting these problems, several directives at EU-level were established to safeguard biodiversity and valuable natural biotopes. For example, under the *Habitats Directive* (1992) the European member states are required to identify and designate Special Protection Areas which are important habitats for the protection of species covered by the directive. Within this directive wetland habitats receive a special status.

In this study we use wetland *restoration* as generic term. This includes not only an improvement in degraded wetlands, but also *re-creation* on sites where similar habitat formerly occurred as well as wetland *creation* in areas where wetlands are established for the first time - within historical time span (MORRIS ET AL. 2006).

Over the last decade, rising political demand for bioenergy in the context of climate change mitigation policies has posed an additional obstacle to ecosystem

preservation and restoration. NILSSON ET AL. (2007), for example, found very large bioenergy resource potentials for Poland. Bioenergy demands increase the value of land and thus, increase the opportunity costs for protected nature areas. As land rents rise, designing space and property for nature conservation has grown to a critical economic and social issue without ignoring production land uses. Protected areas cannot be sustained in isolation from the economic activities in and around them. It is of importance that humans are considered as part of the environment and not only as the underlying problem (LINDENMAYER & HOBBS 2007). Socio-economic considerations and temporal restrictions limit the realization of a chosen restoration goal for a certain wetland or parts thereof. The evaluation of the socio-political interests also includes cost analyses, because all conservation and restoration options incur costs. However, costs have not received much consideration in designs aimed at expanding reserve networks in broader scales (NEWBURN ET AL. 2005).

The principle in the presented study is to optimize different land uses to allow for the persistence and reintroduction of ecosystems by considering bio-geophysical as well as socio-economic factors. This way we can demonstrate the tradeoffs between obtaining higher levels of a conservation target and the increase in cost necessary to obtain it. An important research question is also the potential influence of biomass supply on wetland restoration efforts. The analysis of this study has been executed in European scale by using the EU-25 countries, because conservation planning at broad scales can help to identify areas or regions in which the payoff for conservation efforts is likely to be greatest (WIENS 2007). So far conservationists have mainly focused on finer scales. But there are increasing requests among scientists for embracing and engaging conservation planning at broader spatial scales to obtain a holistic view of the landscape (FRANKLIN & SWANSON 2007; SCOTT & TEAR 2007; WIENS 2007). It is recommended more and more often that the scale of the goals and objectives must also match the scale of the challenge. This implies that a good deal of conservation action must be directed at the scale of land use and of socio-political interests.

7.2. Methodology

7.2.1. The Spatial Wetland Distribution (SWEDI) model

Before evaluating the economic wetland potentials the total wetland area per country needs to be determined. Because of missing base data a methodology to identify wetland distributions including their area potentials has been developed for this study. This resulted in the SWEDI model (CHAPTER 6). The SWEDI model estimates the spatial distribution of European wetlands by distinguishing between existing wetlands and wetland restoration sites. Five wetland types (bog, fen, alluvial forest, swamp forest, wet grassland) are differentiated. SWEDI is a GIS-based model that relies on multiple spatial relationships. It covers the whole EU-25 area excluding Malta and Cyprus at resolution of 1 km². Geographical and physical borders of different wetland types are well reproduced by the SWEDI model as an accuracy assessment with RAMSAR data on selected wetland sites revealed.

The model also differentiates between six wetland size classes, and assesses the restoration success of a potential wetland restoration site after area quality and potential natural wetland vegetation (cf. CHAPTER 8). The results of the SWEDI model were aggregated to country level by maintaining their accuracy in details. Table 7.1 documents its outcome concerning the wetland types. Wetland types of the wetland restoration sites are allowed to overlap because often the wetland type depends on the successional vegetation state and build biotope complexes. A clear separation is neither useful nor tenable in these cases.

In all EU-25 countries, the total restoration potentials amount to 82.5 mio ha and by far dominate the existing wetland areas of about 15.7 mio ha. In Ireland the share of potential existing to the total wetland potential (existing wetlands + restoration sites) is with 26% highest, whereas most countries only show marginal existing wetland areas in comparison to their potentials.

Table 7.1 Country aggregated SWEDI data (in 1 000 ha).

a. Existing wetland areas per wetland type and country

Country	Peatland	Wetforest	Wet-Grassl.	total
Aust	29.65	66.15	0.68	96.48
Belg	11.72	9.90	0.25	21.88
Czec	8.25	77.11	0.39	85.75
Denm	51.40	9.90	2.58	63.88
Esto	186.91	393.56	6.49	586.96
Finl	2 231.91	1 760.87	0	3 992.79
Fran	82.51	492.74	8.81	584.07
Germ	136.28	455.36	13.74	605.38
Gree	25.06	45.39	36.80	107.26
Hung	100.68	299.46	78.14	478.28
Irel	1 162.17	124.01	30.62	1 316.80
Ital	18.62	179.67	18.48	216.77
Latv	152.09	55.02	0.15	207.25
Lith	56.29	22.04	0	78.33
Luxe	0	97	0	0.10
Neth	33.77	47.88	10.85	92.50
Pola	106.98	563.85	0.15	670.97
Port	9.79	57.50	8.36	75.64
Slvn	8.55	5.99	0.06	14.59
Slvk	4.59	46.33	0.97	51.88
Span	66.60	164.75	65.24	296.59
Swed	2 937.68	934.11	5.15	3 876.93
UK	1 423.72	354.26	470.62	2 248.60
total	**8 845.22**	**6 262.85**	**2 080.21**	**15 769.68**

b. Wetland restoration sites per wetland type and country (in 1 000 ha)

Country	Peatland	Wetforest	Wetgrassl.	total
Aust	301.04	196.89	175.31	425.23
Belg	539.48	541.23	118.55	632.74
Czec	596.92	587.05	313.29	819.20
Denm	409.19	414.08	281.20	689.15
Esto	2 682.17	58.21	1 223.29	3001.56
Finl	8 569.11	339.77	12 448.27	16 474.34
Fran	5 118.12	2 939.47	2 241.04	6 836.74
Germ	4 203.85	4 398.47	2 494.23	6 383.74
Gree	797.47	105.75	175.56	886.31
Hung	1 087.72	1 212.59	16 679.25	2 470.16
Irel	1 386.53	406.99	1 229.42	2 171.43
Ital	1 278.66	277.71	558.31	1 720.42
Latv	3 984.41	912.29	891.84	4 350.08
Lith	1 452.77	945.56	674.07	2 005.32
Luxe	17.97	18.81	1.09	24.08
Neth	2 437.53	2 540.10	457.96	2 683.82
Pola	7 850.70	7 863.86	3 333.67	10 154.42
Port	374.42	60.37	159.57	535.73
Slvk	318.26	338.07	182.80	395.79
Slvn	114.13	90.25	38.51	143.18
Span	2 165.02	201.34	660.94	2 659.48
Swed	1 093.35	368.73	8 339.62	8 796.84
Unik	7 362.83	4 783.36	1 274.12	8 294.67
total	**54 141.64**	**29600.89**	**53 951.91**	**82 554.44**

7.2.2. EUFASOM Scenarios

We used the European Forest and Agricultural Sector Optimization Model (EUFASOM) (SCHNEIDER ET AL. 2008) to compute the competitive economic potential of wetlands. EUFASOM is a dynamic, partial equilibrium model of the European Agricultural and Forestry sector, which has been developed to analyze economic and environmental impacts of changing policies, technologies, resources, and markets (SCHNEIDER ET AL. 2008). Land management choices link land, labour, water, forests, animal herds, and other resources to food, fibre, timber, and bioenergy production and their markets. The land management choices include explicitly all major arable and dedicated energy crops, all major livestock categories, more than twenty tree species and forest types, and alternative management systems regarding soil tillage, irrigation, crop fertilization, animal feeding and manure management, and forest thinning.

The geographically explicit resolution of EUFASOM involves member states within EU-25 plus eleven international regions which cover the entire earth. For each EU member state, additional spatial variation can be integrated implicitly via area shares. These differences include a) natural variations pertaining to altitude, soil texture, and slope, b) variations in the state of land and forests pertaining to soil organic carbon levels, forest type, and forest age and c) variations in enterprise structure pertaining to farm size and farming type. However, current computational restrictions do not allow a simultaneous representation of all the above listed differences. The temporal resolution of EUFASOM comprises 5-year periods starting with the 2005-2010 period and terminating anywhere between 2005-2010 and 2145-2150. Exogenous data on state and endowment of resources, land management options and processing technologies, commodity demand, and policies can be adjusted for each period to reflect different development scenarios.

EUFASOM is a large mathematical programming model, which maximizes the discounted sum across regions and time periods of consumer surplus from all final commodity markets plus producer surplus from all price-endogenous resources minus costs for production and commodity trade plus terminal values of standing forests plus benefits from subsidies minus costs from taxes. Restrictions depict resource qualities and endowments, technical efficiencies, crop rotation constraints, environmental impact accounts, political quotas, and intertemporal

relationships for forest inventories, soil organic matter levels, dead wood pools, and timber commodity stocks. The non-linear objective function terms are stepwise approximated to allow EUFASOM to be solved as linear program. Each individual model solution yields optimal levels for all endogenous variables and shadow prices for all constraints. Particularly, production, consumption, and trade variables determine land use and land use change, resource deployment, environmental impacts, and supply, demand, and trade of agricultural and forest commodities. Shadow prices on supply demand balances for resources and commodities identify resource values and market clearing prices for commodities, respectively. Shadow prices on environmental targets reveal the marginal costs of achieving them.

For this study, we extended EUFASOM by integrating the spatially explicit wetland distribution data from SWEDI. We aggregated all spatial units within each EU member state but preserved habitat type, size, and suitability classifications. In addition, we assumed conversion and maintenance costs coefficients for all wetland restoration efforts. To assess the economic potential and agricultural impacts of wetland protection efforts, we specified different scenarios. In particular, we distinguished a) joint vs. country specific wetland targets, b) protected vs. unprotected status of existing wetlands, c) size dependent vs. suitability dependent representation of SWEDI data in EUFASOM, d) wetland targets with vs. without simultaneous European bioenergy target, and e) 20 different wetland targets covering the whole range from no protection to maximum protection. Each selected combination of these scenario assumptions corresponds to a separate solution of EUFASOM.

7.3. Empirical Results

Through EUFASOM scenarios economic and environmental impacts of changing policies, technologies, resources, and markets are analysed to find the socially optimal land use allocation. By including the wetland data into EUFASOM the economic potentials of wetlands, its effects on agricultural and forestry markets, and environmental impacts of wetland protection/restoration efforts are determined for different policy scenarios. Figure 7.1 shows the economic and technical potential of wetlands.

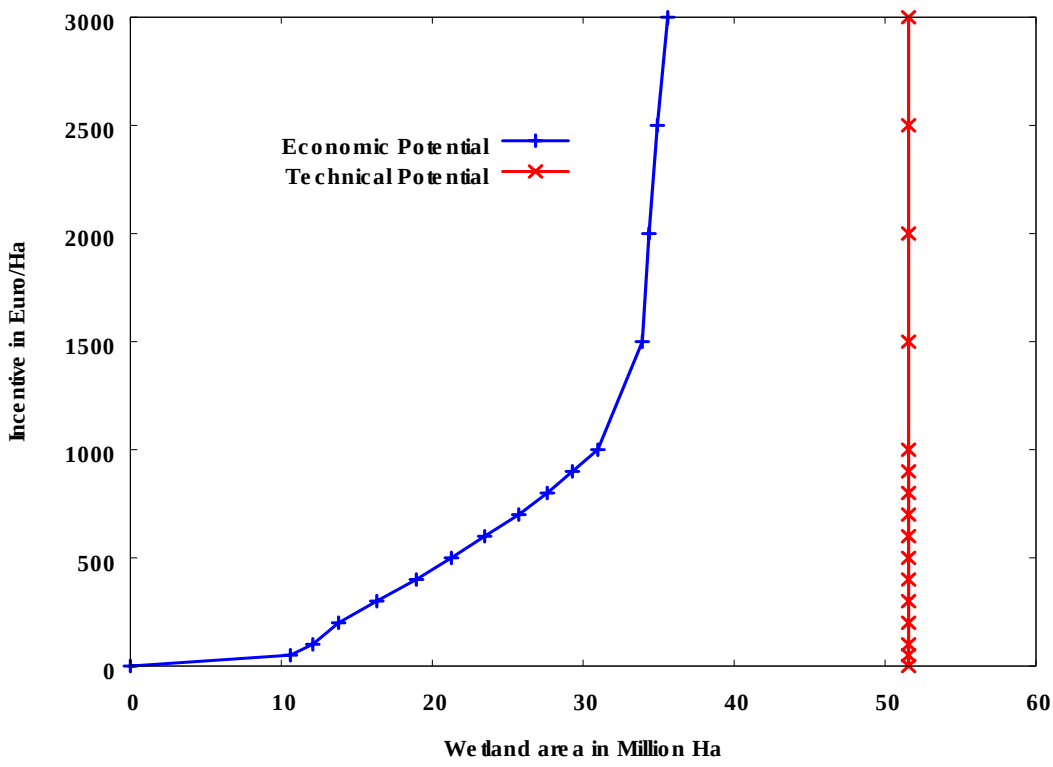


Fig 7.1 Economic versus technical potential of protected wetlands in EU25.

The red curve illustrates the maximum technical potential of wetland area in Europe. Generally, it applies that the more wetland is restored the more expensive the conversion costs become because the marginal costs, i.e. opportunity costs, rise. We included 20 different wetland targets from no protection to maximum protection expressed through incentives in € per hectare converted wetland area. The comparison of economic potential with the technical potential shows that from a certain point on - in this case at incentives of about 1 500 €/ha - additional wetland conversion gets economically unfeasible. As a consequence, the technical potential outreaches the economic potential.

Wetland potentials and its targets are expressed through incentives. As in figure 7.2 shown can these be considered for each country specifically or combined for all EU-25 states. Both curves differ from each other: In the national scenarios wetland restoration targets in one country stimulates agricultural production in other countries due to market linkages. By adding up all national scenarios we achieve an artificial leakage curve as shown in orange at figure 7.2. At EU-25 wide scenarios (blue curve) all countries have the same wetland targets and land competition rises. The bias between the two curves is defined as market leakage. This leakage phenomenon reflects in the differences of the national and EU-25

wide curves this way that the national scenarios have more potential wetland area at lower incentives than the EU-25 wide scenarios (Figure 7.2 panel a).

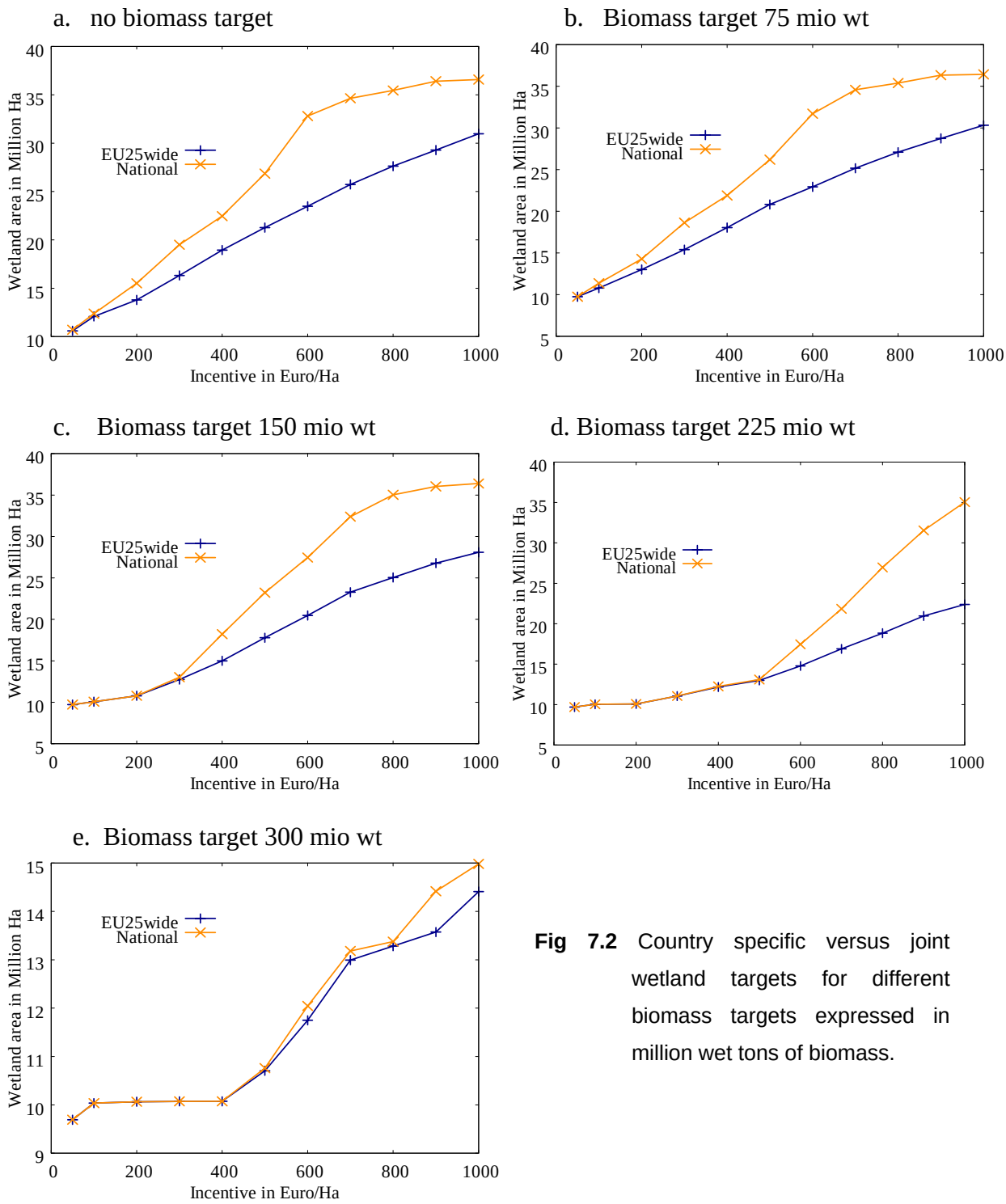


Fig 7.2 Country specific versus joint wetland targets for different biomass targets expressed in million wet tons of biomass.

This means that wetland conversion costs at national incentives are lower because agricultural production may leak to other EU-member states.

The scenarios at figure 7.2 are additionally differentiated after biomass targets of 0 to 100%. The European Union described bioenergy targets for the year 2010 that involves a share of renewable energy of 21% of the total electricity consumption as well as 5.75% bio-fuels of the total fuel consumption. This target can be fulfilled by a supply of about 300 mio. wet tons of biomass. Comparing the national with the EU-25 wide scenarios under consideration of the biomass targets one observes not only a decline in wetland area potentials. Also the national curve reconciles to the EU-25 wide curve the higher the biomass target is set by starting at lower incentives. At biomass target of 100% both curves almost align because the national wetland targets are outweighed by the biomass targets. Consequently, the inclusion of a third component, the biomass targets, into the model resulted in a reduction of the accounting error caused by the national scenarios.

Other scenarios revealed that the establishment of restored wetlands will have impacts on the food price. At figure 7.3 food prices, expressed through the Fisher Index, were integrated into the analysis. Shown are scenarios without wetland protection. In this case, the food prices even fall below 100 as unprotected wetland area is converted into agricultural utilization. The curves show also a dependency on wetland incentives, whereas the “national” scenarios result in lower food prices than the EU25 wide scenarios. The lower the biomass targets the lower are also the food prices due to less competition in utilization demands. The national scenarios with a biomass target of 100% keep clear distance to the other national targets, but show in comparison to the EU-25 wide scenarios hardly a rise in prices even at higher incentives. Again, the leakage factor is visible. At the national scenarios additionally needed food is imported from other countries without wetland targets, whereas at the EU-25 wide scenarios economic costs rise due to competing utilization demands between traditional agriculture, bioenergy plantations and wetland targets. This explains the rise in prices for food.

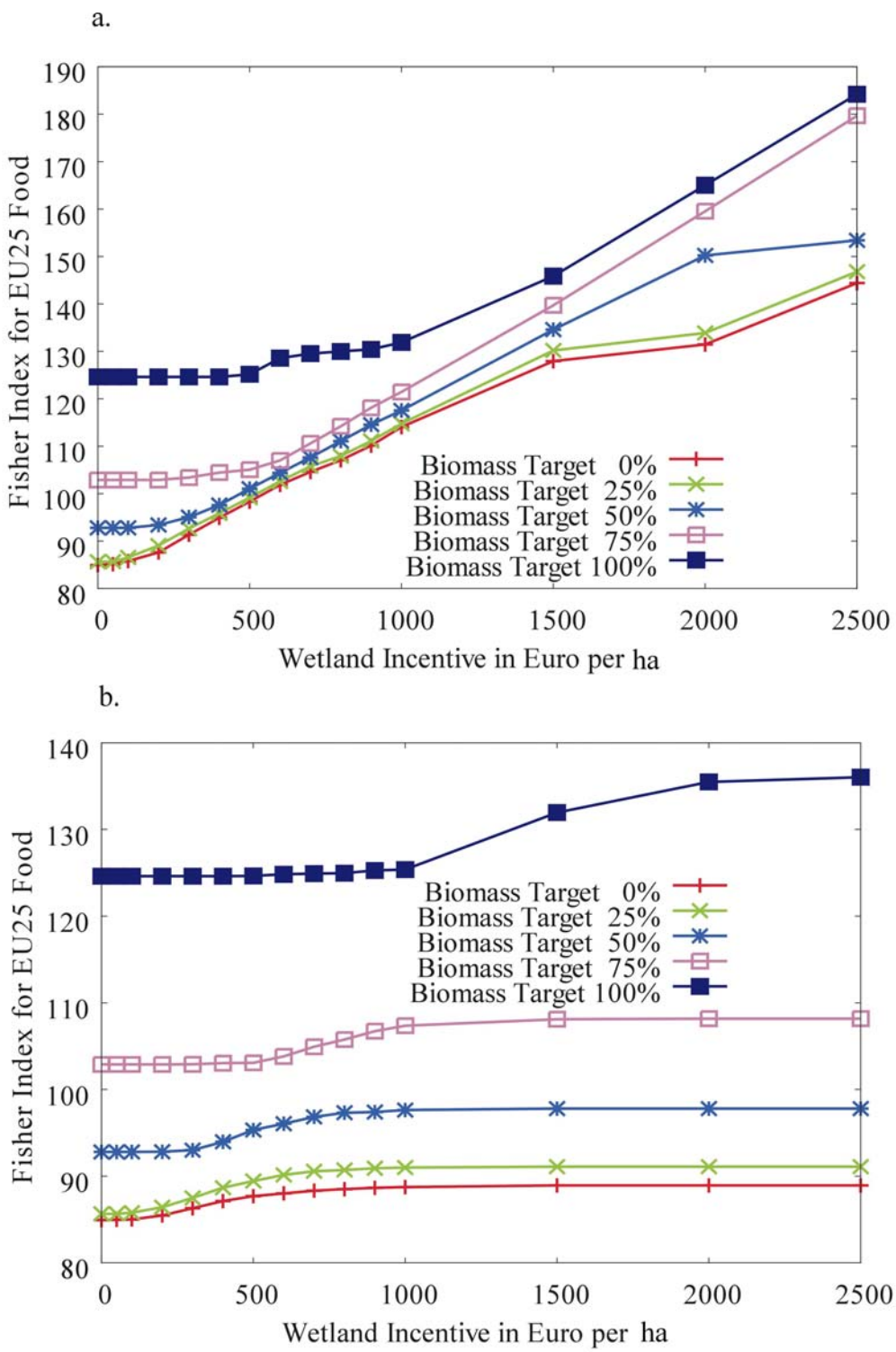
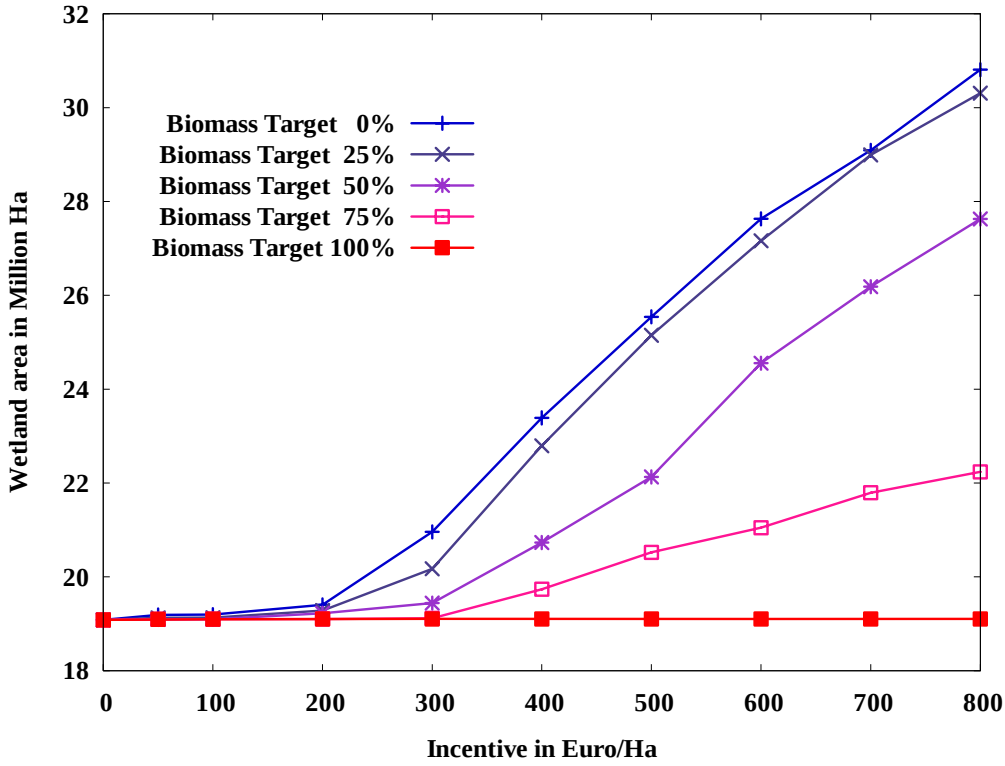


Fig 7.3 EU-25 wide (a) and national (b) scenarios analysing food prices of the EU-25 states in relation to wetland restoration incentives.

EU-25 wide scenarios with joint incentives for all EU-countries are used for the following scenarios. Figure 7.4 distinguishes between protected (a. NoX) versus unprotected status of existing wetlands (b. ALL).

a.



b.

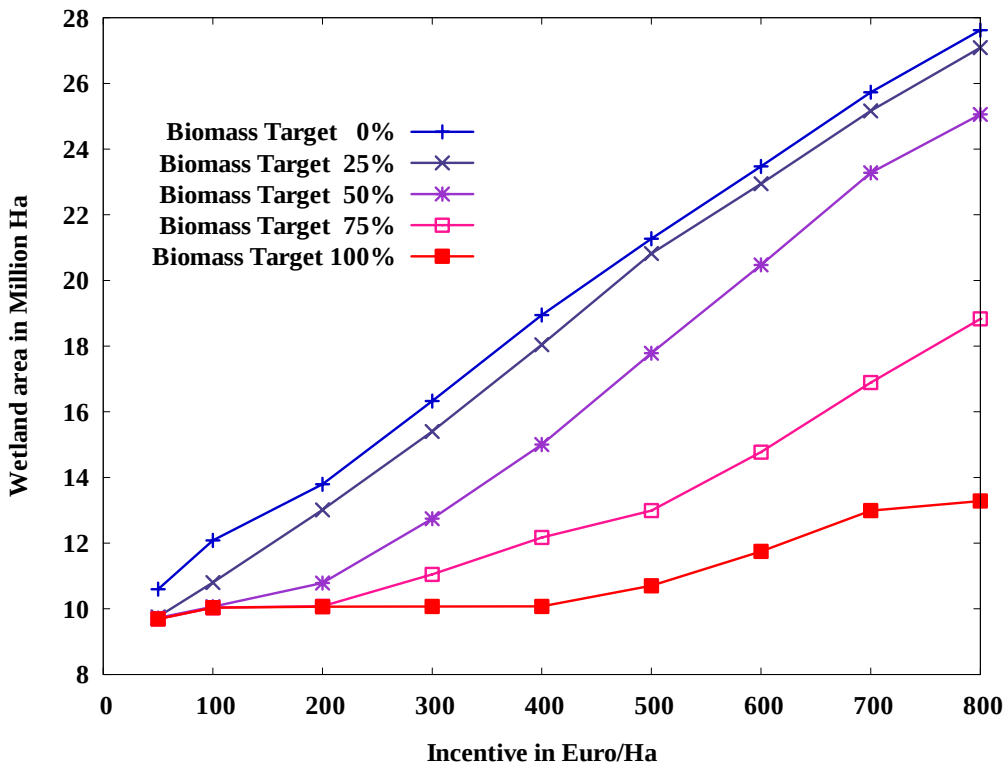


Fig 7.4 Protected (a) versus unprotected (b) status of existing wetlands for different biomass targets.

The protection of existing wetlands implies that these wetlands are not available to be used as agricultural fields or forests, for example, whereas at unprotected status these wetlands may be used for other utilizations as well (cf. figure 7.3). The curves show clear differences also due to different values at the beginning. The scenario with unprotected existing wetlands indicates a more intense rise of wetland area at low incentives, but it also starts at small wetland area in comparison to the protected status, where a rise in wetland area is initiated only from incentives of 200 €/ha. At biomass target 100 even no rise happens at all. The protection-scenarios therefore imply that if wetlands would not be protected, most of the biotopes would be converted into other utilization. Only at incentives of about 400 €/ha the wetland area at scenarios without biomass target reaches the starting point of existing wetland area at protected status.

The EUFASOM scenarios in figures 7.2 to 7.4 show the integration of bioenergy targets with realisation of 25, 50, 75 and 100 % as well as without such target. The results show that in all scenarios biomass targets for climate change mitigation have enormous effects on wetland conservation and restoration. In the following we are going to use the scenarios of figure 7.2 for a more detailed analysis. In this case we chose the EU-25 wide curves of wetland area potentials without biomass target (Fig 7.2.a) and with biomass target 100% (Fig. 7.2 e). We show exemplarily for both cases the wetland potentials for each country separately at incentives of 0, 1000, and 3000 €/ha. Figure 7.5 represents maps of the total potential wetland area per country. It illustrates great wetland potentials at the starting point in Sweden, Finland, but also in the United Kingdom. At this stage the total wetland potentials in Ireland, Poland as well as in Estland are also remarkable, whereas other countries like Italy, Greece, but also Denmark or the Netherlands only have minor total wetland potentials. Comparing now the wetland potentials per country at incentives of 1000 Euro per hectare with and without biomass target one gets only one another picture: The wetland potentials remain stable with biomass target 100%, but the wetland potentials without biomass target show most extending rise in wetland area in France, but also the wetlands in Spain, Germany and Hungary grew as well as wetland areas in Austria, Italy and Greece increased.

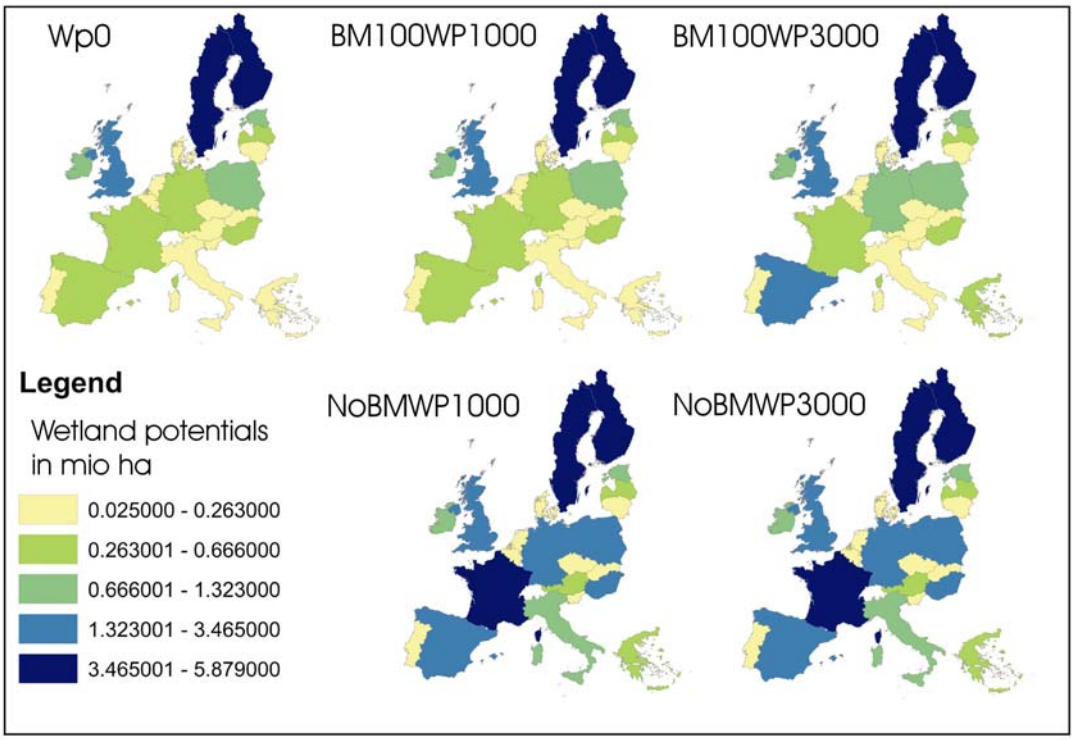


Fig. 7.5 Total potential wetland area per country at incentives of 0, 1000, and 3000 €/ha (WP) with (BM100) and without biomass target 100% (NoBM).

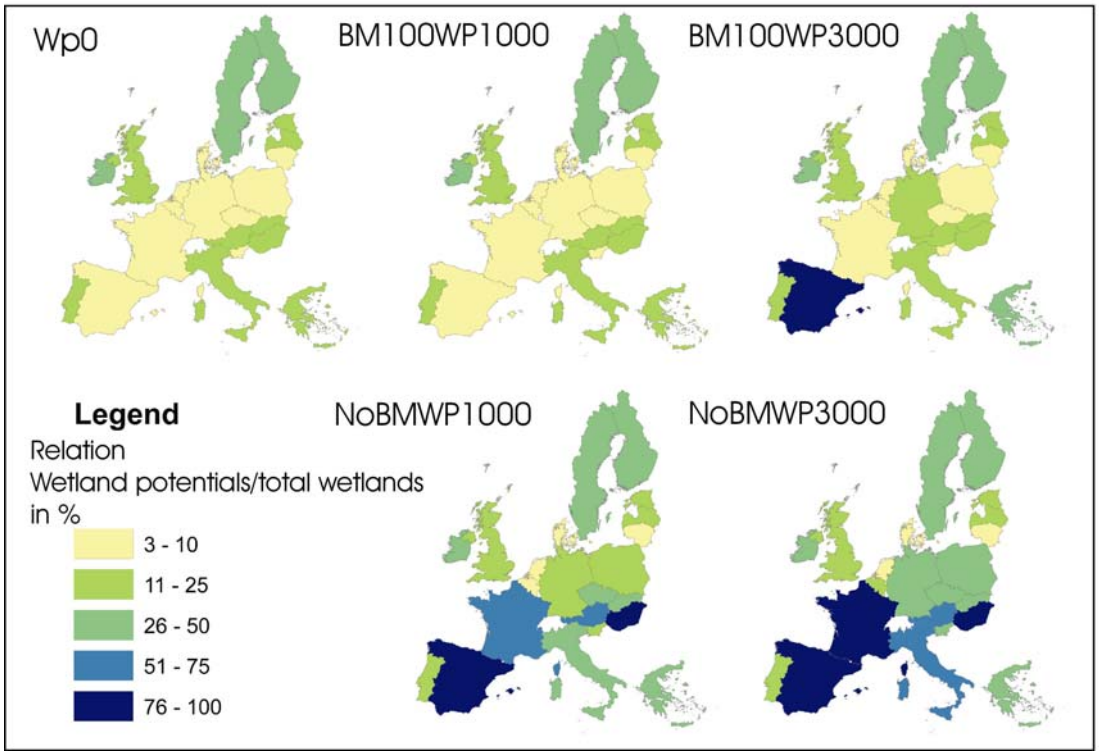


Fig. 7.6 Relation of potential wetland area to the maximum wetland area per country in percent with incentives of 0, 1000, and 3000 €/ha with and without biomass target 100%.

Even if an increase in wetland area took place as figure 7.6 illustrates are the changes in wetland potentials not visible on the map. Therefore shows the map at incentives of 3000 Euro per hectare no differences to the scenarios of 1000 Euro per hectare incentives without biomass target. On the other hand are at the stage of 3000 Euro per hectare increasing wetland potentials at scenarios with inclusion of biomass target 100% visible. In these cases the wetland areas of Spain, Germany and Greece rise considerably.

In contrast to figure 7.5 illustrates figure 7.6 the share of the respective wetland area in relation to the maximum wetland area in percent depending on the EUFASOM scenarios explained through maps. In comparison to the results of figure 7.5, France, Poland and Germany only own minor shares of their total wetland potentials at the starting point, whereas now Italy, Greece, Austria, Slovakia and also Portugal have higher relative wetland area compared to their total wetland area. The maps change drastically at the 1000 Euro incentive without biomass targets where besides the above mentioned countries also the Czech Republic shows rising wetland potentials. The results of the 3000 Euro incentives without biomass target indicate that the share of wetland potentials to the total potential wetland area of France, Germany, Poland and Italy increased more than in other countries. The high shares of 76 to 100% of Spain or France, for example, results from relatively small total wetland potentials of that country.

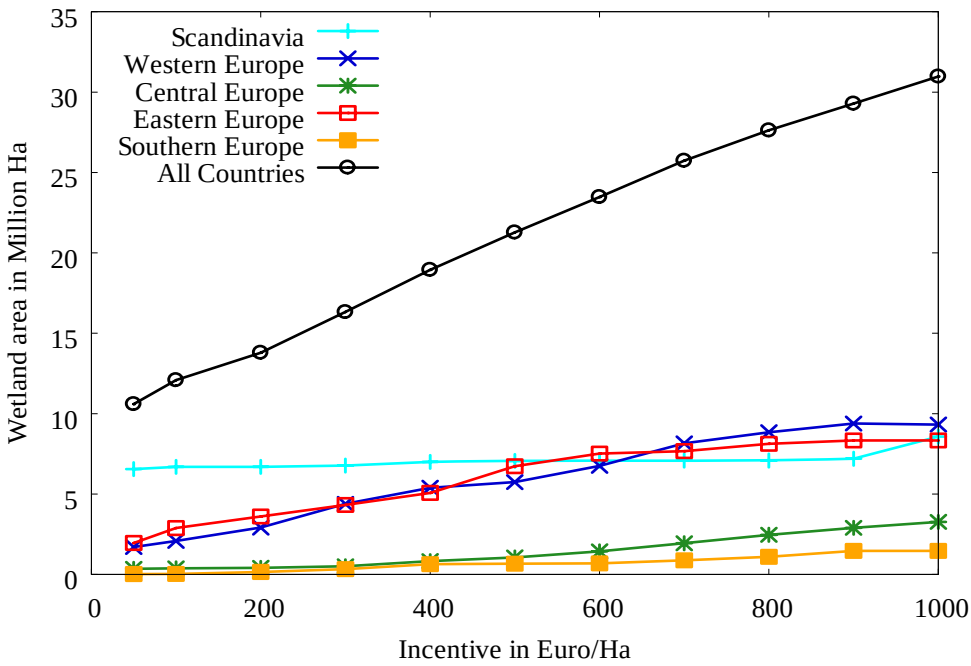
To learn now more about regional differences we aggregated the data of the potential wetland areas into regions (Table 7.2) to illustrate potential differences in wetland potentials.

Table 7.2 Definition of EU-25 regions.

Regions	Countries
Scand	Finland, Sweden
East	Estonia, Hungary, Latvia, Lithuania, Poland, Slovakia
Central	Austria, Belgium, Czech Rep., Denmark, Luxembourg, Netherlands
West	France, Ireland, Portugal, Spain, UK
South	Greece, Slovenia, Italy

Figure 7.7 illustrates these differences in more detail by comparing scenarios with biomass target 100% and without biomass target.

a.



b.

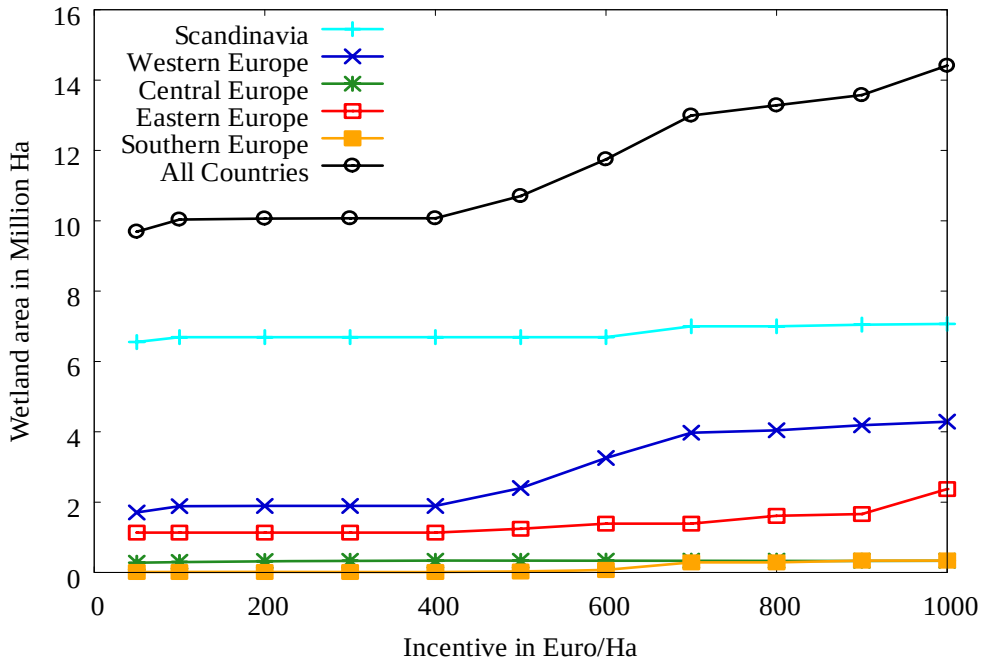


Fig 7.7 Regional distribution of unprotected wetlands for a. biomass target 100% and b. Biomass target 0.

The wetland area in the Scandinavian Region keeps nearly constant independent of biomass targets. By far the most extending wetland increase is observed in

Western European where the wetland area even raises above the Scandinavian wetland potentials at the scenario without biomass target. Here, also the Eastern European region shows extending growth in wetland area similar to the Western European region. This is not the case at scenarios with biomass targets. The Central and South European regions show an increase in wetland area only at scenarios without biomass targets. In relation to their low total wetland potentials due to geo-ecological factors the share in rise of wetland area can be even rated higher than elsewhere in this case.

7.4. Summary and Conclusions

The GIS-based SWEDI model estimates the spatially explicit distribution of existing and potential wetlands. Results show not only a heterogeneous distribution across countries but also large differences between the two areas. Potential wetland areas in Europe are about five times larger than existing wetlands. To evaluate the competitive economic potential of wetland preservation under different policy options, SWEDI data were aggregated and integrated into EUFASOM. This bottom-up, partial equilibrium model portrays the competition for scarce land between agriculture, forestry, dedicated bioenergy enterprises, and nature reserves. Production intensities, prices, international trade, and demand for agricultural and forest commodities are endogenous. As shown in chapter 7.3, the spatial extent of wetland preservation is sensitive to incentives. It is relatively inexpensive to achieve moderate levels of conservation but marginal cost rise steadily as the total protected areas increase (ANDO ET AL. 1998; POLASKY ET AL. 2001; NAIDOO & ADAMOWICZ 2005). Note that incentives of several thousand Euro per hectare are easy to simulate with a mathematical programming model but difficult to realize politically.

Wetland targets in one place stimulate land use intensification elsewhere due to market linkages. Thus when wetland restoration in one country reduces agricultural production the market is likely to cause this to be offset by increased production elsewhere (cf. GAN & MCCARL 2007). This leakage phenomenon indicates also that environmental stresses, in this case to wetlands, may be transferred to other countries (cf. BRUVOLL & FÆHN 2006). However, we find that wetland conversion rises when a national rather than an EU-25 wide perspective is

employed. On the other hand reduces the introduction of biomass targets the bias between national and EU-25 wide perspectives due to additional land utilization demands.

Large wetland areas impact food production, consumption, and market prices. Higher food prices rise the opportunity costs of wetlands. If these cost changes are ignored, the resulting marginal cost predictions can be substantially underestimated. Similarly, adding nationally obtained cost estimates understates the true cost of EU-wide preservation incentives. In independent national assessments, costs appear lower because agricultural cost changes from simultaneous preservation policies in other countries are neglected.

Existing European wetlands are relatively well protected through EU-policy measures. However, these areas may need to be extended to realize the ambitious political targets related to biodiversity protection.

Bioenergy targets have enormous effects on conservation planning and nature conservation. An enforcement to achieve the EU-bioenergy target, meaning to produce about 300 mio wet tons of biomass per year, would lead to less wetland restoration areas at very high incentives, but even to no additional wetlands, respectively conservation areas, than the existing at incentives up to 1000 Euro per hectare. This also reflects in regional and country-specific analyses.

Regional and country-specific differences in wetland potentials exist as well. The wetlands are not evenly distributed due to their geo-ecological and spatial relationships but also because of economic aspects like land costs, for example.

The presented study helps to find the socially optimal balance between alternative wetland uses by integrating biological benefits – in this case wetlands - and economic opportunities – here agriculture and forestry. The analyses offer insights into environmental conservation effects in European scale caused by policy driven land use changes. Spatial data provide a possibility to build the interface between economic and ecologic models.

8. A cost-efficient site-selection model for European wetland restoration

8.1. Introduction

In this study, we concentrate on the area potentials to preserve, restore or create freshwater wetland ecosystems in the European Union in consideration of economic and bio-geophysical aspects. Expansion of settlements, agricultural areas and bioenergy plantations at the expense of wetlands and its corresponding fragmentation constitute a great challenge to nature conservation. Therefore, the understanding of how spatial patterns influence ecological processes at land use scale level as become an important factor in landscape management (EHRLICH 2007). Even though wetlands constitute valuable ecological resources, the number and size of wetlands in Europe has dramatically decreased over the last century. Main areas of wetland conversion include agriculture, forestry, peat extraction on fens and bogs, as well as urbanization and infrastructure measures (JOOSTEN & CLARKE 2002). Fens and floodplain forests have been opened up culturally since the early Middle Ages, but their major decrease has happened during the last few decades of the 20th century (RAMSAR COMMISSION), when private profit maximizing land use decisions resulted in drainage of wetlands and degradation. The diverse utilization demands lead to direct biotope loss and habitat fragmentation of the remaining wetlands in Europe. In spite of important progress made in recent decades, wetlands continue to be among the world's most threatened ecosystems, owing mainly to ongoing drainage, conversion, pollution, and over-exploitation of their resources.

Over the last decades concerns to the consequences of wetland degradation have been rising. Because the large-scale destruction of wetlands causes not only ecological damages but also negative economic externalities, as heavy floods in the vicinity of regulated rivers often illustrate. Subsequently, several conventions and directives and with them a range of natural conservation and restoration action have been adopted for the protection of wetlands (e.g. Natura 2000 sites, Water Framework Directive, Ramsar Convention). In this study, the term

restoration includes an improvement in degraded wetlands as well as re-creation on sites where similar habitat formerly occurred and also wetland creation in areas where wetlands are established for the first time within historical time span (MORRIS ET AL. 2006). Restoration and conservation management are increasingly viewed as complementary activities with restoration often now forming an important element of conservation management (YOUNG 2000; HOBBS 2005). The reason is that the size and structure of existing reserves are often inadequate to provide certain biodiversity benefits. It is therefore necessary to acquire additional land with habitat value or restoration potential (MILLER 2007). Ecosystem restoration has therefore become a vital tool in the maintenance and restoration of resilience (MANNING 2007) even if wetland restoration effect is debated vehemently (cf. MORRIS ET AL. 2006; RATTI ET AL. 2001; ZEDLER & CALLAWAY 1999; ZEDLER ET AL. 2001; HOBBS 2007). However, wetland regulations should be designed to conserve an array of wetland functions, and not be limited to water quality, waterfowl habitat or recreation. They should simultaneously address all major functions, and connectivity of wetland, aquatic and terrestrial resources, and be comprehensive enough to protect both, individual wetlands and the overall integrity of landscapes in which wetlands occur (CALHOUN 2007).

The value of restored wetlands depends on its size, structure, and the surrounding landscape (MARTIN ET AL. 2006; MC INTYRE 2007). Values increase if protected areas are integrated into wider landscape uses and are connected to other areas of similar qualities. During the last years, the emphasis of conservation has shifted from protecting species to preservation of entire ecological systems or functional landscapes (WIENS 2007). Thus, efficient conservation policies must take the landscape context and function into consideration (HOBBS 2007, WESTPHAL ET AL. 2007), where humans are considered as part of the environment and not only as the underlying problem (LINDENMAYER & HOBBS 2007). Especially in Europe, their influence on the environment over many thousand centuries should not be neglected. Different abiotic, biotic and landscape specific cultural interactions and conditions have led to the characteristic spatial heterogeneity in Europe (HABER 1979). A strategic coordination is important to achieve greater benefits, such as by integrated networks of habitat (BENNETT & MACNALLY 2004). The most appropriate targets for restoration and the most cost-effective means of achieving

clearly stated goals should be evaluated. This includes that the ecosystems would be able to interact with current surrounding landscapes as well as that the solution would be accepted by human societies (NILSSON ET AL. 2007). The key question of this study is therefore what pattern is most suitable to achieve an effective habitat network across the landscape for target ecosystems, in this case wetlands.

The aim of this study is to develop a decision support tool that uses spatially explicit land-use data to identify priority areas for wetland preservation considering both ecological linkages at the landscape level and full costs under different policy scenarios. To achieve this aim, we use a landscape approach to determine an EU-wide wetland network that 1) gives priority to the preservation of existing wetlands over restored wetlands, 2) includes the value of connectivity among these wetland systems and processes, 3) facilitates the ability of the wetlands and its surroundings to function as dynamic systems, 4) allows the biota of these ecosystems and landscapes to adapt to future environmental changes (HOCTOR ET AL. 2000), and 5) accounts direct and opportunity costs of preservation.

Protected areas cannot be sustained in isolation from the economic activities in and around them. Resources available for conservation management have always been limited. In this context is it essential that management actions are prioritized and directed towards explicitly stated goals and targets. Socio-economic considerations and temporal restrictions limit the realization of a chosen restoration goal for a certain wetland or parts thereof. One aim of this study is to incorporate costs into the spatial wetland site selection model to demonstrate the tradeoffs between obtaining higher levels of a conservation target and the increase in cost necessary to obtain it. It is relatively inexpensive to achieve moderate levels of conservation but often quite expensive to achieve maximum levels (ANDO ET AL. 1998; POLASKY ET AL. 2001; NAIDOO & ADAMOWICZ 2005). In the past, costs have not received adequate consideration in designs aimed at expanding reserve networks (NEWBURN ET AL. 2005). In reality, economic constraints impinge upon any landscape planning or design problem, and different assumptions about economic costs can result in markedly different solutions. For landscape restoration, the economic costs would include acquisition costs, management costs, transaction costs, and opportunity costs (NAIDOO ET AL. 2006;

WESTPHAL ET AL. 2007). We use the European Forest and Agricultural Sector Optimization Model (EUFASOM) (SCHNEIDER ET AL. 2008) to compute the corresponding economic potential of wetlands, its effects on agricultural and forestry markets, and environmental impacts for different policy scenarios. EUFASOM is a partial equilibrium model of the European Agricultural and Forestry sector, which has been developed to analyse changing policies, technologies, resources, and markets (SCHNEIDER ET AL. 2008). The main purpose is to make possible consistent analysis of abatement cost curves for greenhouse gas emissions, and how changing policies, technologies and market conditions influence these costs. The model is scaled at EU country level but considers variation in natural conditions within countries. EUFASOM's objective function maximizes total agricultural and forestry sector surplus.

The methods and mechanisms by which wetland restoration sites could be identified differ (cf. BURNSIDE AT AL. 2002). Often habitat suitability models determine the required habitat content or context for single or multiple species and restore landscape accordingly (VILLARD ET AL. 1999; PRESSEY ET AL. 1997; HAIGHT ET AL. 2004; WESTPHAL ET AL. 2007). The underlying methodology mainly relies on weighted scoring approaches where rankings for each attribute are used to calculate the geometric mean as a measure of overall suitability (HOCTOR ET AL. 2000; BURNSIDE ET AL. 2002; TREPEL & PALMERI 2002). Another approach to model site-selection is the construction of decision-modelling frameworks, as described for example in POSSINGHAM & SHEA (1999) and POSSINGHAM ET AL. (2001). Several site-selection studies utilise GIS to map the modelled geographic distribution of individual species (POWELL ET AL. 2005; BAYLISS ET AL. 2005, CHEFAOUI ET AL. 2005). However, all of these studies rely on the modelling of environmental envelopes of one or multiple selected species and not on whole differentiated ecosystems as represented in this study by wetlands. LONKHUYZEN ET AL. (2004) evaluated the suitability of potential wetland mitigation sites using GIS and TREPEL & PALMERI (2002) modelled the nitrogen retention of wetlands at landscape scale. They state that the success of wetland restoration is dependent on the site-selection to achieve specific restoration goals. The aim of the spatially explicit wetland site-selection model

presented here is to allow a flexible modelling process that is able to accommodate these different and multiple planning goals simultaneously.

8.2. The wetland site-selection model – a methodological introduction

The wetland restoration site selection model is part of an integrated modelling system, which comprises three main components:

The first component is a spatially explicit GIS-based distribution model of Europe (SWEDI, CHAPTER 6) with a spatial resolution of 1 km² that uses several spatial data (Corine land Cover, European Soil Database, Bioclim, Worldclim, Gtopo30, and Potential Natural Vegetation) and its combination concerning specified wetland characteristics. It differentiates between existing wetlands and potential restoration sites. SWEDI currently distinguishes five wetland types.

The highly resolved wetland areas of the SWEDI model are upscaled to EU country levels and passed to the second component, the European Forest and Agricultural Sector Optimization Model (EUFASOM, SCHNEIDER ET AL. 2008). EUFASOM is used to estimate the economic wetland potential expressed in hectare wetland area per EU-country and wetland type. The model is a fully dynamic, partial equilibrium model with endogenous commodity prices. Possible land exchanges and competition between agriculture, forestry, bioenergy, and nature reserves are represented. EUFASOM can be subjected to different policy settings, technological progress assumptions, and environmental change scenarios.

The third component involves a GIS-based site-selection model, which downscales the country-based, scenario specific results from EUFASOM to a higher spatial resolution. In the following exposition, we focus only on the geo-ecological development of the site-selection model. Figure 8.1 gives an overview of the methodological structure.

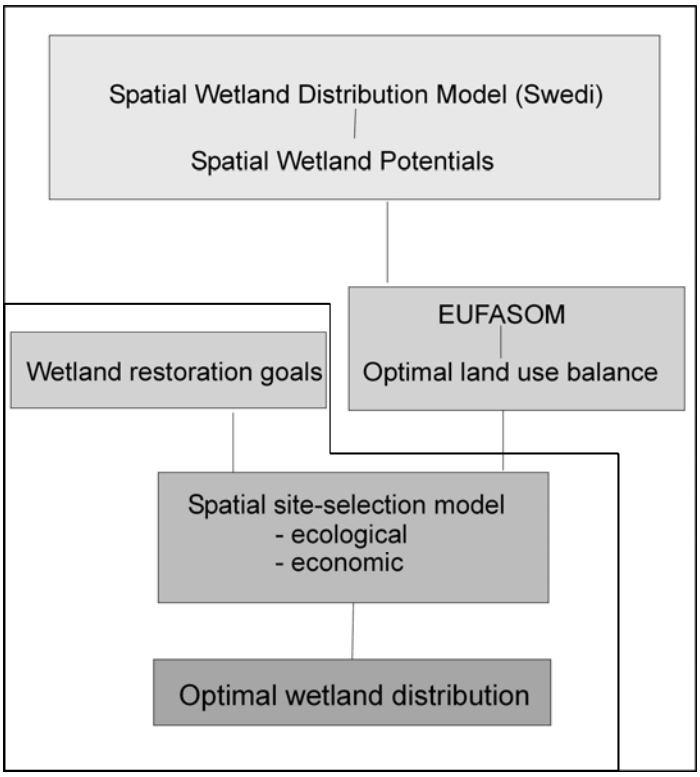


Fig 8.1 Overview of the methodology

The wetland location is often important in terms of ecological functions and values to people. These wetland functions and values do not only depend upon size, shape, type, and other characteristics of a wetland, but also upon proximity and connections with other waters, water quality, adjacent buffers, threats, and a broad range of other factors (KUSLER & KENTULA1990).

Site selection also depends on the specified goals. For the spatially explicit modelling of optimal wetland distribution it is necessary to formulate goals to identify and prioritise potential wetland restoration sites (KUSLER & KENTULA 1990, HOBBS 2007; SCOTT & TEAR 2007). These objectives may differ between regions, countries, or wetland types. The combination of several objectives or targets depending on its country and on the wetland type makes the formulation of a number of scenarios possible. The pre-defined potential goals may be applied separately as single objective but also combined in multiple objectives. In the following table (8.1), we define the potential ecological targets for the site-selection analysis as well as list its underlying evaluation methods that are explained below.

Table 8.1 Environmental goals and its evaluation methods

Restore the potential wetland sites that	**Evaluation method**	**abbr.**
lie within certain range to existing wetlands/conservation areas	spatial join	**DIST**
are directly attached to existing wetlands/open waters/rivers/lakes	spatial join	**Att**
build biotope complexes; improve connectivity among existing wetlands	proximity	**PX**
enlarge existing wetlands to a certain size	spatial join, area	**En**
are of certain size	area	**A**
are potential convertible to peatland/wetforest/wetgrassland sites	wetland type	**W**
lie on extensively used grassland/arable farmland/forest	land use	**LU**
are ranked after geophysical site-suitability	suitability assessment	**SUIT**
are prioritized after area quality	hemeroby assessment	**AQ**

Landscape metrics are the basis for the detection of each goal's spatial distribution. For each goal, a spatially distributed land attribute is calculated (TREPEL ET AL. 2000). Due to scale, these landscape attributes are more explanatory than patch-specific metrics. The analyses are conducted using *ArcGIS* as well as the analysis tools *V-late* and *Hawths Analysis Tools* (2006, TIEDE 2005, LANG & TIEDE, 2003). The wetland distribution of the SWEDI model is used as input parameter (CHAPTER 6). This model allows a detailed wetland type distinction in European scale for existing wetlands, but also for potential restoration sites. The other parameters used for the wetland site-selection assessment are extracted from CORINE Land Cover 2000 data (EEA 2000). In the following, the restoration goals are described in more detail:

a. Distance (DIST). Areas that lie within a certain range of existing wetlands or of conservation areas are detected by applying the spatial join function of ArcGIS and setting the match distance to the preferred range. The potential wetland restoration sites are in this case spatially joined with the existing wetland sites. All potential wetland restoration sites that fall - fully or partly - within this match distance are selected.

DIST = PCS *spatial join* (PEH *match distance* X)

b. Attachment (Att). Wetland restoration sites that are directly attached to existing wetlands or open waters are evaluated by using the spatial join function as well (see a.). However, the match distance is set to a minimum of 10 m. 10 m are selected instead of 0 to allow for spatial or geometrically uncertainties in the SWEDI model.

Att = PCS *spatial join* (PEH *match distance* 10)

c. Proximity. The proximity index (PX) rates individual wetland patches according to its functional network with the surrounding wetland habitats (KIEL & ALBRECHT 2004). It analyses isolation or complexity of biotopes by distinguishing between space dispersal and clustered distribution of biotopes by considering the size as well as the distance of the patches (GUSTAFSON & PARKER 1992).

$$PX = \sum_{i=1}^{n} \frac{A_i}{d_i}$$

PX is calculated for patch *i* of a certain wetland class that is totally or partially situated within the defined proximity buffer. A_i is the patch size, and d_i is the nearest neighbor distance to a patch of the same class within the selected buffer.

The distribution of wetland sites of the SWEDI model serve as input for the PX evaluation and build the base of the subsequent PX scenario analyses of potential wetland restoration sites to allow comparability. The search radius is set to 2 km. To obtain reasonable results, the PX value needs to be transformed logarithmically (based on WEIS 2007). The PX decreases the smaller the area and/or the higher the distance to similar patches of land becomes. The value is highest if a patch is surrounded by and/or extending towards nearby biotopes of the same kind (LANG & BLASCHKE 2007). Table 8.2 shows the classification scheme of the PX for existing wetlands.

Table 8.2 Classification scheme

log (PX)	Description
-4.193 – -0.3497	small isolated wetlands that are not able to connect to other wetland systems
-0.3498 – 0.5905	spatially isolated wetlands of moderate size
0.5906 – 1.408	small to moderate wetlands with only moderate importance for connectivity
1.409 – 2.389	extending but spatially isolated wetlands or wetlands of any size that serve as important stepping stones between other wetlands
2.390 – 8.442	extending wetlands that build complexes with other wetlands

In a second step, the potential convertible sites were added to the existing wetland sites to repeat the PX evaluation under above described conditions. All wetland restoration sites with a PX above two thirds of the defined base PX of the existing wetland areas are assumed to reach the selected goal by building complexes.

d. Enlargement (En). Another goal is to enlarge the existing wetlands to a certain size. In this case the potential wetland restoration sites are selected through the attachment analysis (see b). Subsequently, the suitable areas of a defined minimum and maximum area of the combination of existing and selected convertible sites are evaluated by using SQL queries.

En = ((PCS spatial join PEH match distance 10) < x) AND ((PCS spatial join PEH match distance 10) > Y)

e. Size (A). The larger the habitat the less it is influenced by its surroundings and the higher is the probability for the establishment of viable populations (WULF 2001). For this reason, potential wetland restoration sites of a certain size may be selected. In this case, the desired minimum or maximum size of a potential wetland needs to be determined. With the help of SQL statements, the distributions of the potential wetland sites can be determined.

A = (PCS > X) AND (PCS <Y)

f. Wetland type (W). The Swedi model distinguishes five wetland types and six structures. Through SQL queries, the desired wetland types can be selected from the potential wetland restoration sites.

$$W = W_{P,Wf,G}$$

g. Land use (LU). Corine Land Cover data are used to identify the land use on the potential wetland restoration sites. The model is able to prioritize certain land use classes.

$$LU = X_{LU}$$

h. The suitability assessment. The wetland site suitability is assessed based on its potential restoration success and mainly dependent on ecological information. In addition, the surrounding land use, topography, and water quality can influence both technical and economical feasibility and hence the long-term success of a constructed wetland (PALMERI 2002). We use the wetland distribution of the SWEDI model (CHAPTER 6). The model assumes that the current land use on the potential wetland restoration sites and in its surroundings plays an important role in the restoration success. “Suitable” wetland restoration sites are further assessed with regard to its land use quality. The current land use at and around suitable sites is determined through Corine land cover 2000 data. Urban and other sealed off areas and their direct vicinity are assumed to be unsuitable for wetland restoration. All potential wetland restoration sites that fall within urban or other artificial areas including a 500 metre buffer are therefore extracted from the model. Furthermore, those sites that contain already existing conservation areas like salt marshes or valuable sparsely vegetated areas are also excluded as potential wetland restoration sites. Remaining potential restoration sites fall within agricultural areas and forests. Within these areas, wetland suitability is assessed by intersecting the potential wetland sites with extracted areas of potential wetland vegetation of the Potential Natural Vegetation map of Europe (BFN 2004). It is assumed that wetland restoration sites that match the potential natural wetland vegetation would be more easily restored than other sites which involve less conversion and management costs. Therefore, those potential wetland restoration sites that fall within the PNV wetland area are considered “suitable”, whereas the remaining potential wetland restoration sites are considered “marginal”. Figure 8.2 shows the distribution of suitable and marginal potential wetland restoration sites per country.

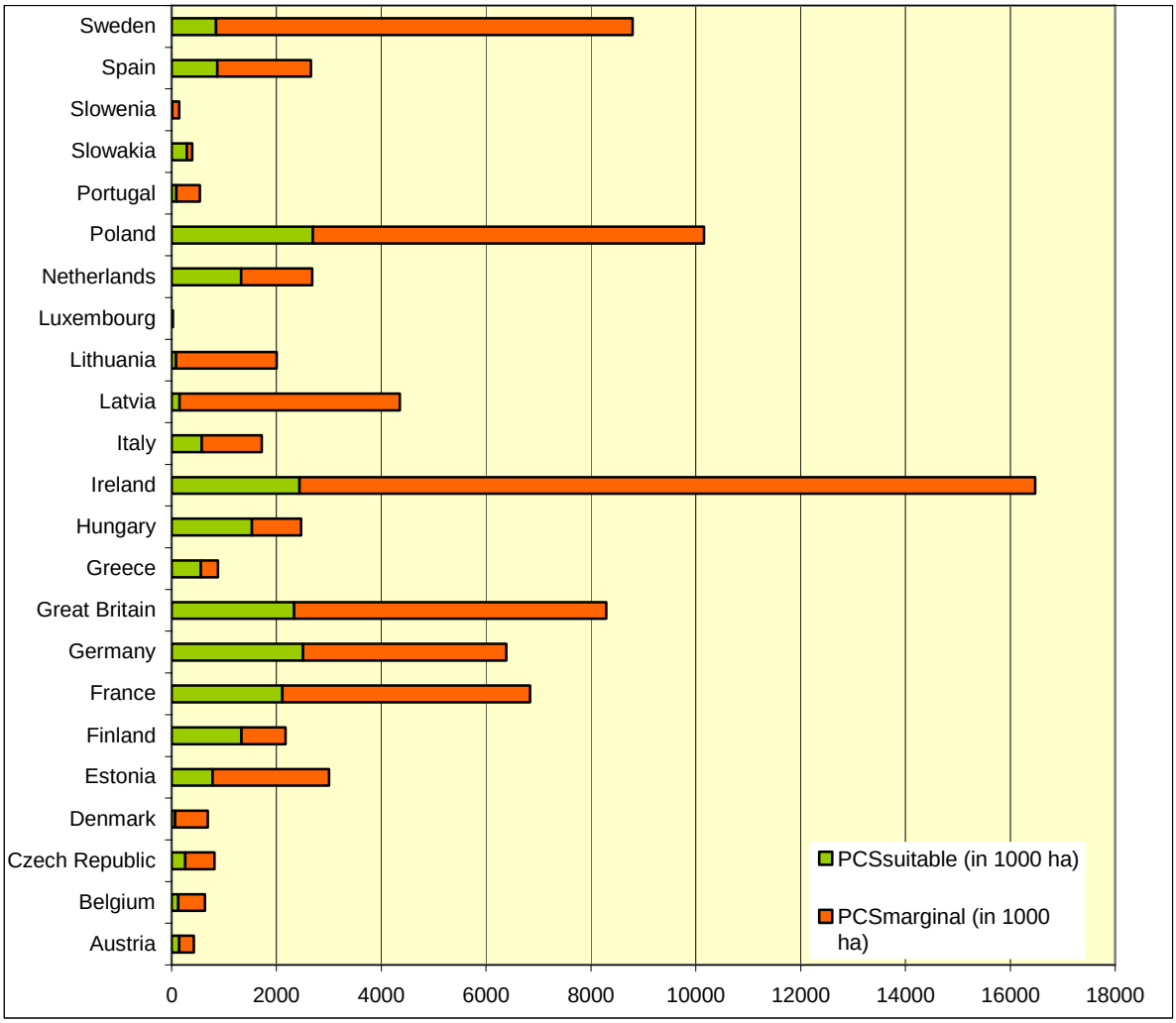


Fig 8.2 Results of the site-suitability assessment.

About two thirds of the potential sites for wetland restoration yield “marginal” sites. In Ireland, Greece, Hungary and Slovakia, “suitable” potential wetland sites dominate over the “marginal” sites. The Netherlands achieve nearly as many suitable sites as marginal ones. In comparison to the other EU-25 countries, France, Germany, Great Britain and Finland, as well as Poland have high amounts of “suitable” wetland restoration sites. As shown on the map in figure 8.6, the most suitable conversion sites are found within river valleys, next to open waters and other existing wetlands. In figure 8.6 in the appendix this distribution is illustrated in more detail through a map.

i. Area Quality. Neighbouring land use and the quality of the potential wetland restoration site can influence the long-term success of a constructed wetland.

Therefore, the site-selection model considers the site quality as well as neighbourhood qualities of the surrounding areas of potential wetland restoration sites. Corine Land Cover data (EEA 2000) are assessed considering how close the respecting land use on potential restoration sites and their surroundings is to its original natural state, given the influence of anthropogenic cultivation present. This is expressed through the hemerobic index (*HI*). In general, the ranking follows the assessment by GLAWION (1999; see above). It is mainly based on vegetation but depends directly on human utilization intensity and pressure. The *HI* is closely connected to the biological regulation and regeneration capacity (SCHLÜTER 1987). The lower the *HI* is, the more limited is the regulation and regeneration potential of the biotope. This allows inferences about the ecological stability of assessed landscapes. The *Hawth' analysis tools* (2006) are used to characterize the spatial context around the potential wetland sites within a 1 km neighbourhood (WESTPHAL ET AL. 2007). The area-relevant mean value of the *HI* including the *HI* of the potential wetland restoration site is determined for each patch (see also BASTIAN 1997; SCHLEUPNER & LINK 2007). It is expressed in six classes. Table 8.3 shows the classification scheme of the hemerobic index values of the potential wetland restoration sites and their application to Corine Land Cover data.

Potential wetland restoration sites are situated exclusively on sites with *HI* 2, 3, and 4. Figure 8.7 in the appendix illustrates the distribution of wetlands and their hemerobic ranking and gives an overview of the hemerobic classes applied to the EU-Corine land cover.

Table 8.3 Explanation of the hemerobie-index (*Hem class* after Glawion 1999, changed)

Hem class	Corine Land Cover Classes	Describtion	HI
I	open spaces with little or no vegetation wetlands water bodies	natural close to nature	1
II	forests shrub and/or herbaceous vegetation association	semi-natural	**2**
III	pastures heterogeneous agricultural areas	conditionally far off nature	**3**
IV	arable land, permanent crops	far off nature	**4**
V	mine, dump, and construction sites artificial non-agricultural vegetated areas	artificial	5
VI	urban fabric industrial, commercial and transport units	unnatural	6

Figure 8.7 shows that the largest areas for potential wetland restoration with the highest site qualities are found in Scandinavia, the west coast of Scotland, and the Baltic States. Fragmented sites are found in eastern Germany and Poland as well as in France and Hungary. Medium area qualities appear mainly along the North Sea states, in Ireland but also in the Baltic states. The low quality sites are scattered across entire Europe with the exception of Ireland.

The restoration goals described above are the basis for the spatial site-selection model. Out of these, specific algorithms are combined to integrative and complex statements as described below. Each raster cell of the SWEDI model that has been evaluated as potential wetland restoration site is attributed with the above stated goals. Consequently, one layer is produced for each restoration goal. Single or multiple goals determine the site-selection process. Single-goal selection makes no further analysis necessary, because the spatial model automatically chooses the selected attribute as "suitable" site. If the selected area of the restoration goals (S) exceeds the area determined by EUFASOM, the site-selection model chooses a sub area of the potential wetland restoration sites according to their quality. The area quality can also directly be applied for the site-selection equation. In this case, those sites with higher area quality are prioritised in the rank of its prioritisation goals.

Multiple goals depend on the analysis of logical connections between the layers using Boolean logic. The potential wetland restoration sites that are selected for each of the restoration goals receive a suitability value of "1"; the remaining wetland areas, which don't fall into the categories, obtain a value of "0". The site-selection model also integrates wetland sites of neighbouring countries into the analysis, to avoid adulterating results along the country borders.

For different search criteria, different potential site selection maps can be produced. These maps can be analysed through the summed irreplaceability algorithm (WESTPHAL ET AL. 2007) to identify those sites that are chosen more than randomly by the model in dependence of their EUFASOM wetland potentials. The summed irreplaceability I can be computed for each site i over all wetland restoration scenarios r as follows (after WESTPHAL ET AL. 2007):

$$I_i^{qr} = \frac{\sum_{i=1}^{r} S_i}{p_i^{qr}}$$

where q is the wetland potential based on EUFASOM and p is the probability that site I would be selected at random at a 95 % confidence interval. At normal distribution we assume an equal probability for each site to be chosen depending on the EUFASOM wetland potentials.

8.3. Results

We apply the EUFASOM scenario results shown in figure 8.3 to illustrate the wetland site selection. Wetland potentials are assessed simultaneously with and without European bioenergy targets (cf. CHAPTER 7). The main assumption behind these scenarios is that the European Union has formulated bioenergy targets for the year 2020 that involve a 20% share of renewable energy in its total electricity consumption as well as a 10% bio-fuel share in its total fuel consumption. If the first target would be fulfilled through biomass based electricity, about 300 mio wet tons of biomass would have to be supplied. This would require significant impacts on land use. This is confirmed by EUFASOM scenario results which show that biomass production targets have substantial effects on wetland conservation and restoration potentials. As figure 8.3 illustrates, the wetland restoration area in scenarios without a biomass target, increases steadily for incentives up to 1000 Euro per hectare. On the other hand, a biomass target of 300 mio wet tons makes wetland restoration incentives below 1000 Euro per hectare ineffective. Only at very high incentives, some wetland becomes converted. The wetland potentials computed by EUFASOM scenarios give the optimal wetland area per EU-25 country for each given policy option. This area is then downscaled by optimising several restoration goals to find the most efficient sites to restore. The restoration goals are in this case ecological and geographical parameters determined through landscape metrics and spatial analyses as explained in chapter 8.2.

In the following, we illustrate the downscaling of EUFASOM scenarios of figure 8.3 and their integration into two different multiple restoration goals in more detail by applying the model exemplarily to Germany.

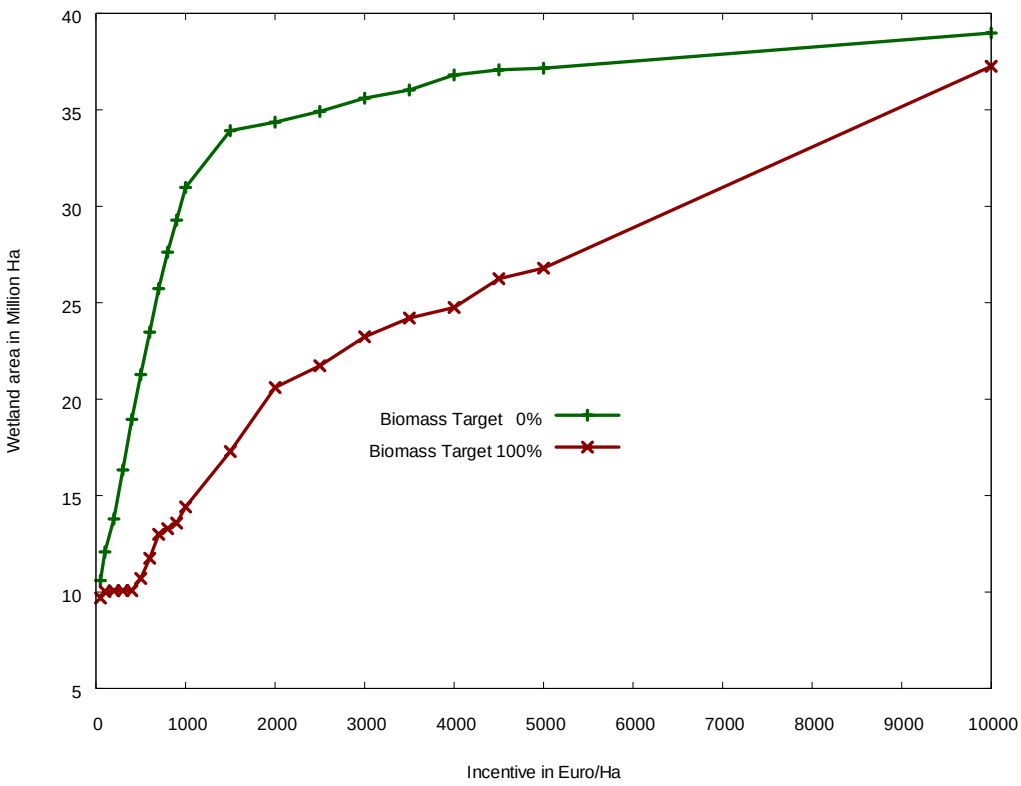


Fig 8.3 Wetland potentials with biomass target 100% and without biomass target.

The multiple restoration goals are not static and always rely upon design and management objectives that might be regionally differentiated as well. In this example, we define the objectives as follows:

> *Select those sites that improve the connectivity among existing wetlands Prioritise areas that are directly attached to open waters. Of these, prefer "suitable" sites over "marginal sites".*

According to the restoration goals, the model chooses the three layers PX, Att, and SUIT and initially combines those using logical connections:

(SUIT) = 1 AND (Att) = 1 AND PX >= 1.4

The challenge is now not only to determine suitable wetland restoration sites in dependence of the restoration goals, but also to connect these with economic wetland potentials of the EUFASOM scenarios. These determine the maximum area of potential wetland restoration per country. For this reason we need to extend the equation with a constraint for each scenario limiting the potential wetland area that varies:

WetlArea (type) (country X) <= Y (scenarioZ)

In case the resulting selection of potential wetland restoration sites exceeds the required maximum wetland potential, the site-selection model prioritizes the sites depending on its area quality (AQ) until the limit is reached. Because the PX changes with altered wetland areas for each individual wetland site, it is recomputed after each modelling step. If the selected wetland area remains below the maximum wetland limit, some criteria for certain parameters can be relaxed; in this case the PX value of 1.4 would be reduced until the wetland limit is reached. The results can be illustrated in dynamic maps depending on the wetland potentials of EUFASOM and restricted to the modelled wetlands of SWEDI. Figure 8.4 shows the results of our exemplary restoration goals for selected wetland potentials at incentives of 1000 and 3000 €/ha (see figure 8.3). In reality, these incentives are more than unrealistic but in this scale they make the differences of site-selection readily observable.

The four maps illustrate the differences in maximum wetland potentials and therefore also show the different distributions of selected wetland areas for restoration. The scenarios shown here assume protection of existing wetlands, so that the area of existing wetlands remains constant and only potential wetland restoration sites are allowed to change in area extent. In scenario A (Biomass target 100%, incentive 1000 €) no additional area is provided for wetland creation. The already very high incentive of 1000 €/ha is not sufficient to compete with the demands from biomass plantations. Without a biomass target the wetland area of Germany would at incentives of 1000 € per hectare even triple its extent to about 1.9 mio hectares. These wetlands are mainly distributed along river courses and in the low lying North Sea coastal region. In both scenarios, incentives of 3000 € per hectare result in a rise of wetland area. However, the wetland potential in scenario B (Biomass target 100%, incentive 3000 €) doubles in comparison to scenario A. It shows similar wetland selection as scenario C (no target, 1000 €) but only with less wetland area. Scenario D (no target, 3000 €) yields the highest wetland potentials of the four cases. According to the restoration goal to enhance connectivity, the potential wetland sites are distributed between other potential wetland restoration sites and consequently enlarge the biotope complexes.

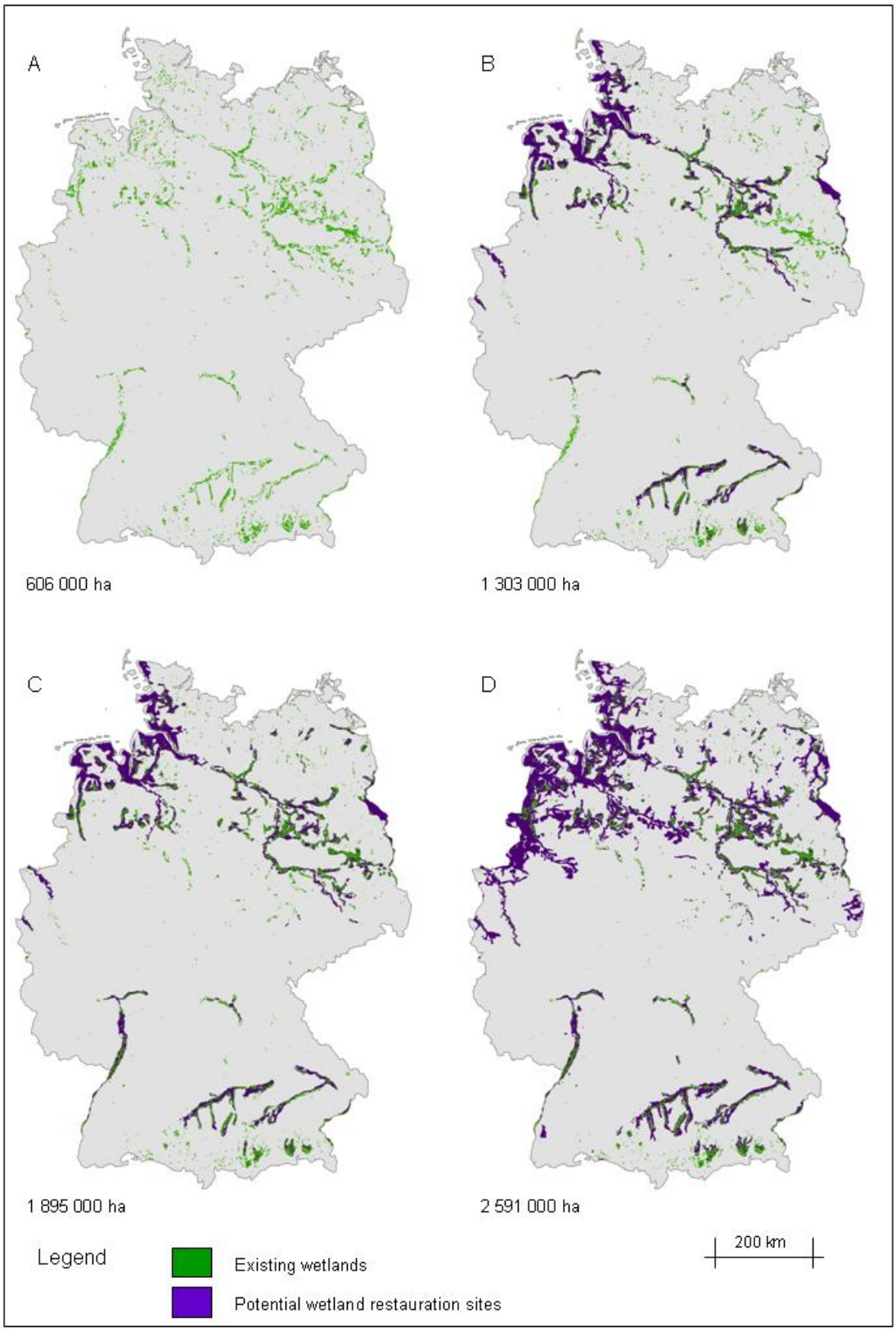


Fig 8.4 Exemplary wetland site selection for the defined restoration goals. A. 1000 € incentive with biomass target 100%, B. 3000 € incentive with biomass target 100%, C. incentive 1000 €/ha without biomass target, and D. 3000 € incentive without biomass target. Numbers indicate the respective maximum wetland potential including already existing wetland areas.

Via multiple combinations of restoration goals several different wetland site-selection scenarios can be obtained with each one showing unique wetland constellations also depending on the EUFASOM wetland potentials. These unique wetland solutions may be used to evaluate the summed irreplaceabilities after WESTPHAL ET AL. (2007). But whereas WESTPHAL ET AL. (2007) apply the irreplaceability algorithm for each scenario and budget size we utilize the summed irreplaceability equation to denote the number how often a site is selected in different scenarios depending on the same budget size after EUFASOM. This consistency is an expression of the priority or importance of the selected sites to be restored because it fulfils most of the restoration goals. Exemplarily we apply the summed irreplaceability algorithm to the wetland potential of 1 895 000 ha in Germany as given in scenario C in figure 8.4 (1000 € incentive without biomass target). Using the restoration goals, we construct 20 different site selection scenarios of wetland restoration. In figure 8.5 the summed irreplaceability is illustrated by a map.

The map visualizes priority sites for wetland restoration through summed irreplaceability analysis. The wetland restoration sites illustrated with red colour on the map are those sites that are chosen less often than the determined average probability. The potential wetland restoration sites with highest rates are found mainly in north-western Germany in river valleys of smaller water courses as the Aller or Leine rivers or the upper river course of the Ems. Other sites with high restoration priority are the river valleys of the Danube and Isar River in southern Germany, but also Elbe and Oder valleys in northern and eastern Germany. Weser and Ems river lower catchments areas show high selection rates as well.

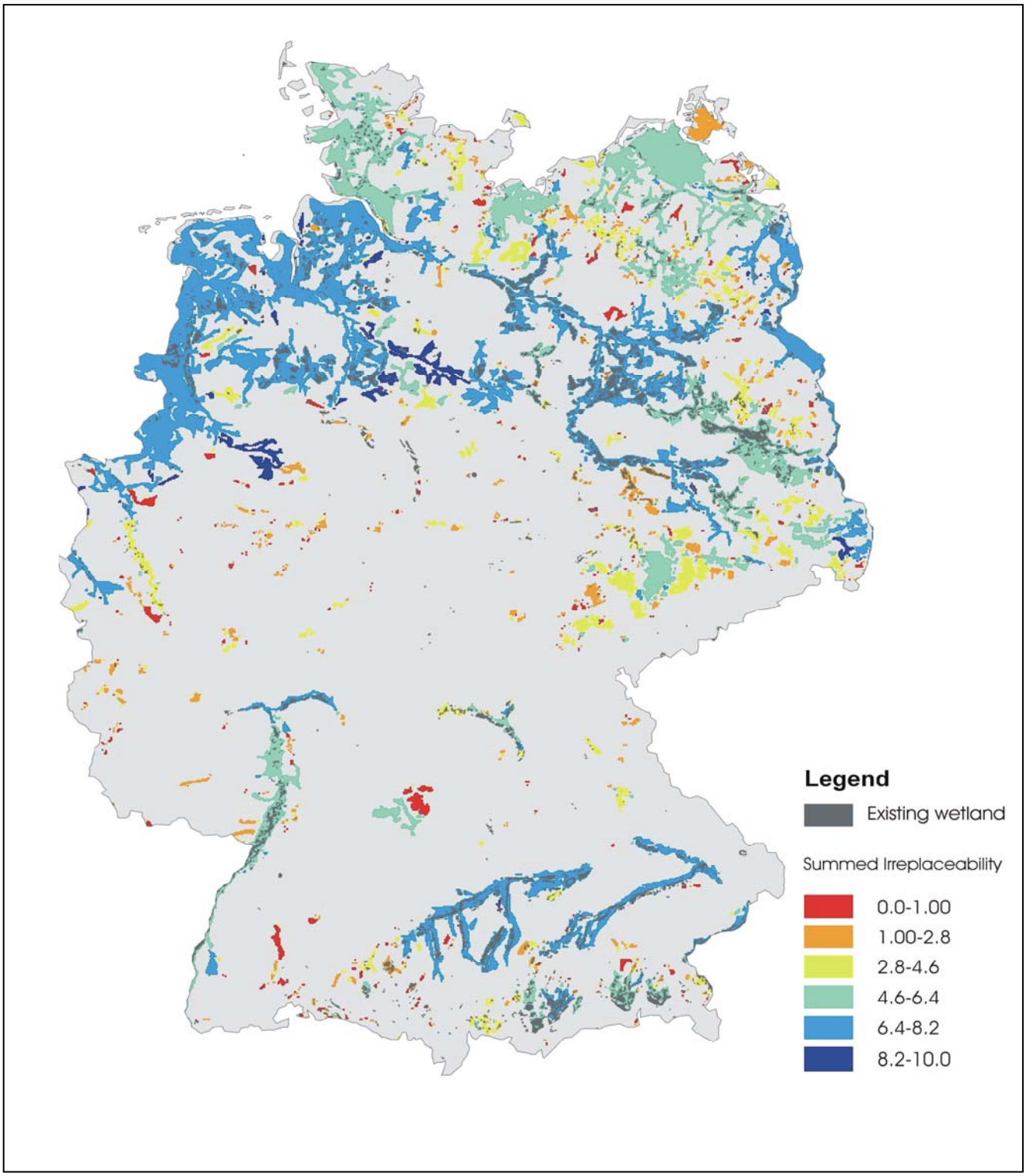


Fig 8.5 Summed irreplaceability of wetland restoration sites of Germany over 20 different scenarios.

8.4. Discussion and Conclusions

Loss of natural ecosystems due to increasing land demands for other purposes is a major threat to many species but also to the sustainable development of the landscape in which the loss occurs. In general, ecosystem degradation constitutes

social costs. This leaves researchers, policymakers, and society with two important questions: i) what degree of preservation is desirable, i.e. socially optimal, and ii) which sites should be chosen for preservation? Conservation planning needs to be weighed against other environmental and societal objectives (SCOTT & TEAR 2007; WIENS 2007). Therefore, conservation planning at global, broad ecoregional scales can help to identify areas or regions in which the payoff for conservation efforts is likely to be greatest (WIENS 2007).

We focus our study on European freshwater wetlands to evaluate the regional potentials of wetland restoration. During the last century more than 60% on average - in some countries even more than 80% - of all European wetlands were drained and converted to other land uses and the loss still continues despite several European conservation efforts. We distinguish between three preservation options: First, existing habitats can be protected from destruction. Second, on suitable sites, habitats can be restored. Third, on both existing and restored habitats, ecological management can increase the suitability and carrying capacity for certain species. Each of these options incurs costs. These costs consist of i) direct costs, i.e. the costs of restoration, maintenance management, and protection, and ii) opportunity costs. Direct costs are low where little restoration and maintenance is necessary. Opportunity costs are low where alternative land uses yield small benefits. Through the integration of the spatial wetland distribution model SWEDI into the economic optimization model EUFASOM it is possible to obtain statements of potential wetland areas per EU-country depending on different policy scenarios. EUFASOM considers also the effects of wetland conservation and restoration on agricultural and forestry markets. The location of the wetland in the landscape strongly influences its function. Knowledge about interrelationships between the wetland and the surrounding landscape is important for the success of wetland restoration projects as well as for the protection of natural, presently undisturbed wetlands (DAVIDSSON ET AL. 2000). Existing wetland areas and wetland restoration sites need to be integrated into wider landscape uses and be connected to other areas of similar qualities. Connected sites improve the survival chances of species in response to disturbances and climate change. The *Natura 2000* network can play a role in achieving such integration (EEA 2004). This study makes a contribution towards these goals by

considering the interaction of natural, engineering, economic and human sciences. The site selection model uses SWEDI data to downscale the results from the economic analysis by utilising landscape metrics that analyse combinations of wetland restoration goals. These targets build the basis variables for the wetland site selection model that uses land-use data and information on significant ecological functions to identify potential ecological linkages. Potential wetland restoration sites might be used as buffer zones between existing wetlands and intensively used areas, they might also be important for the creation of corridors, step stones, or connections to other valuable existing wetlands. As result we obtain a spatially realistic, GIS integrated model (cf. LAUSCH 2004) that shows varying potential convertible wetland sites in the order of their restoration goals and dependent on the EUFASOM scenarios. By using the methodology of summed irreplaceability, we are further able to identify areas of ecological wetland priority and to make statements about large scale wetland conservation targets. The results indicate that wetlands along waterways and with a certain minimum extent are prioritized over smaller and fragmented wetlands. This result highlights the meaning of water-systems for the interconnectedness of greater ecosystems. However, the analysis has been conducted at country scale and therefore the assessment of potential wetland restoration sites as non-priority sites does not make any inferences about their landscape value.

As WIENS (2007) concluded conservation planning at large scales can help to identify the most suitable sites for conservation efforts. However, nature conservation must also be detected at broader scales of land use policies, which has often been neglected. Therefore, this GIS model was developed to depict the optimal distribution of wetlands at coarse geographic scale. This involves integrating a variety of GIS datasets and multiple iterations of interpretation. Despite the great opportunities that large-scale site-selection models offer, one always has to deal with spatial uncertainties and data limitations. In general, it is necessary to know the origin of the inflowing water, flow paths in the landscape and the fate of the water leaving the wetland. Information about the catchments, geology, geomorphology, vegetation and land-use are also needed. But often the available data are very general at this level of scale if available at all. Therefore, this site-selection model is primarily meant to show possible solutions in certain

scenarios, and not to yield particular locations for wetland restoration at small scale. Moreover, it certainly is another challenge to facilitate and realise the technical and site-specific options to restore wetlands on local scale than simply select areas for restoration as done in this study at European scale. The model is useful to locate areas suitable for restoration programs, for the introduction of faunistic corridors considering the *Natura 2000* network, and favouring success in regional conservation planning. Additionally, an associated study is going to integrate area requirements of selected wetland species into the model. The results also give an overview of vital planning information for more detailed regional studies.

The site-selection model is considered to be extended to a decision making tool for identifying areas within a landscape where multiple utilisation demands overlap in geographic space. In a next step also economic constraints and relationships are going to be included as well. This module is in progress but has not provided sufficient base data for introduction into the site-selection model yet. The current model version excludes socio-economic constraints apart from EUFASOM data in terms of spatially explicit costs in the analysis. In future, the spatially explicit wetland site-selection model presented here will consider not only the costs of wetland conservation and restoration but also its spatial variability (e.g. BALMFORD ET AL. 2003; NEWBURN ET AL. 2005). It aims to come to more realistic solutions in optimal reserve design with the help of constraints of economic reality. As the site selection maps show, the most suitable wetland restoration sites are mainly found within river valleys of great agricultural value. Therefore, it is expected that the integration of spatially explicit costs will have enormous impacts on wetland site-selection. The impact of climate change to wetland restoration is another topic that might be investigated after expanding the site-selection model. The site-selection model is an attempt to effectively reduce human threats and biodiversity losses in Europe. The integration of both, optimal wetland conservation options and economic land use allocations within a GIS environment is an important step forward in interdisciplinary cooperation in terms of land use management and the formulation of environmental policies.

8.5. Appendix

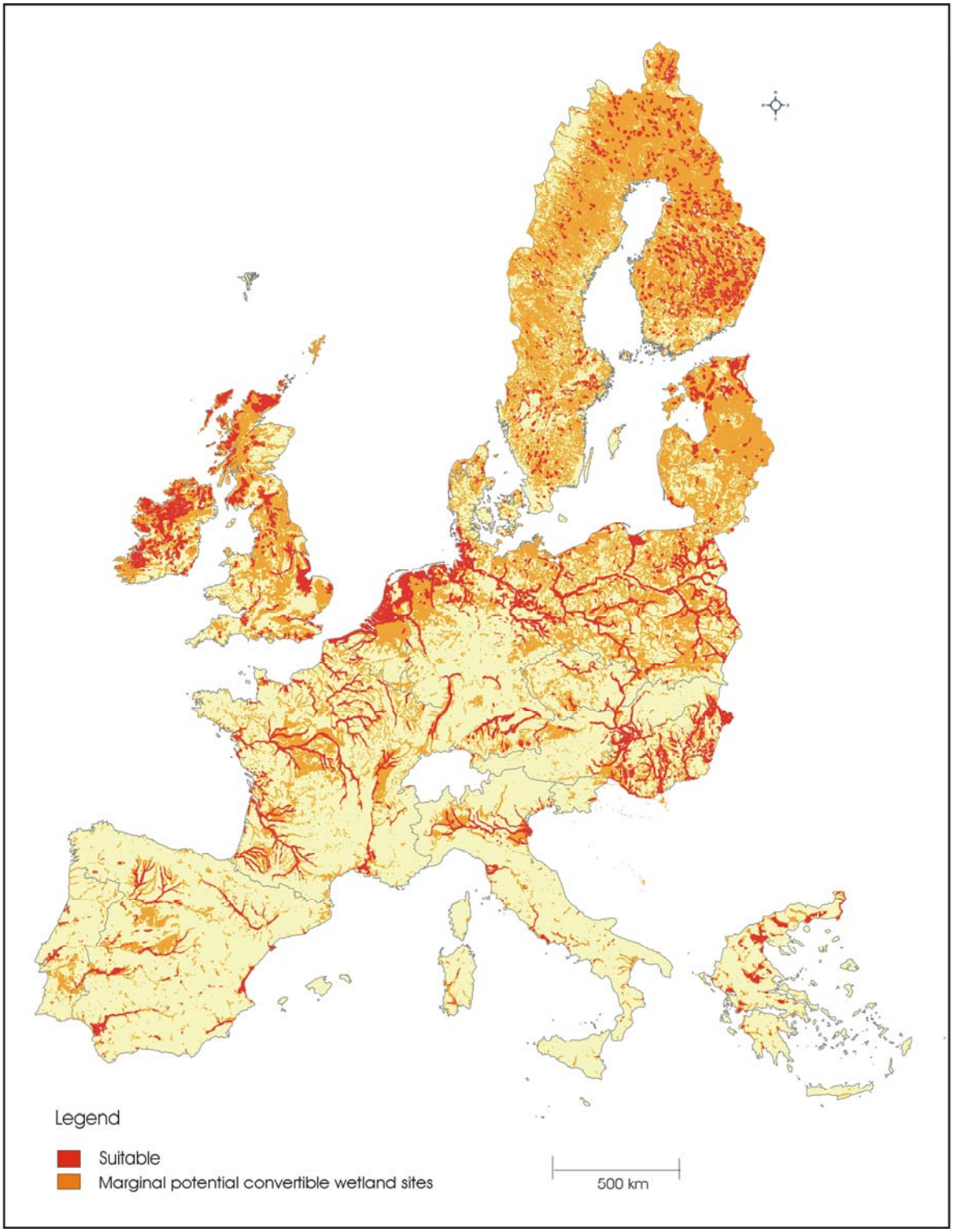


Fig. 8.6 wetland site suitability assessement of potential convertible sites.

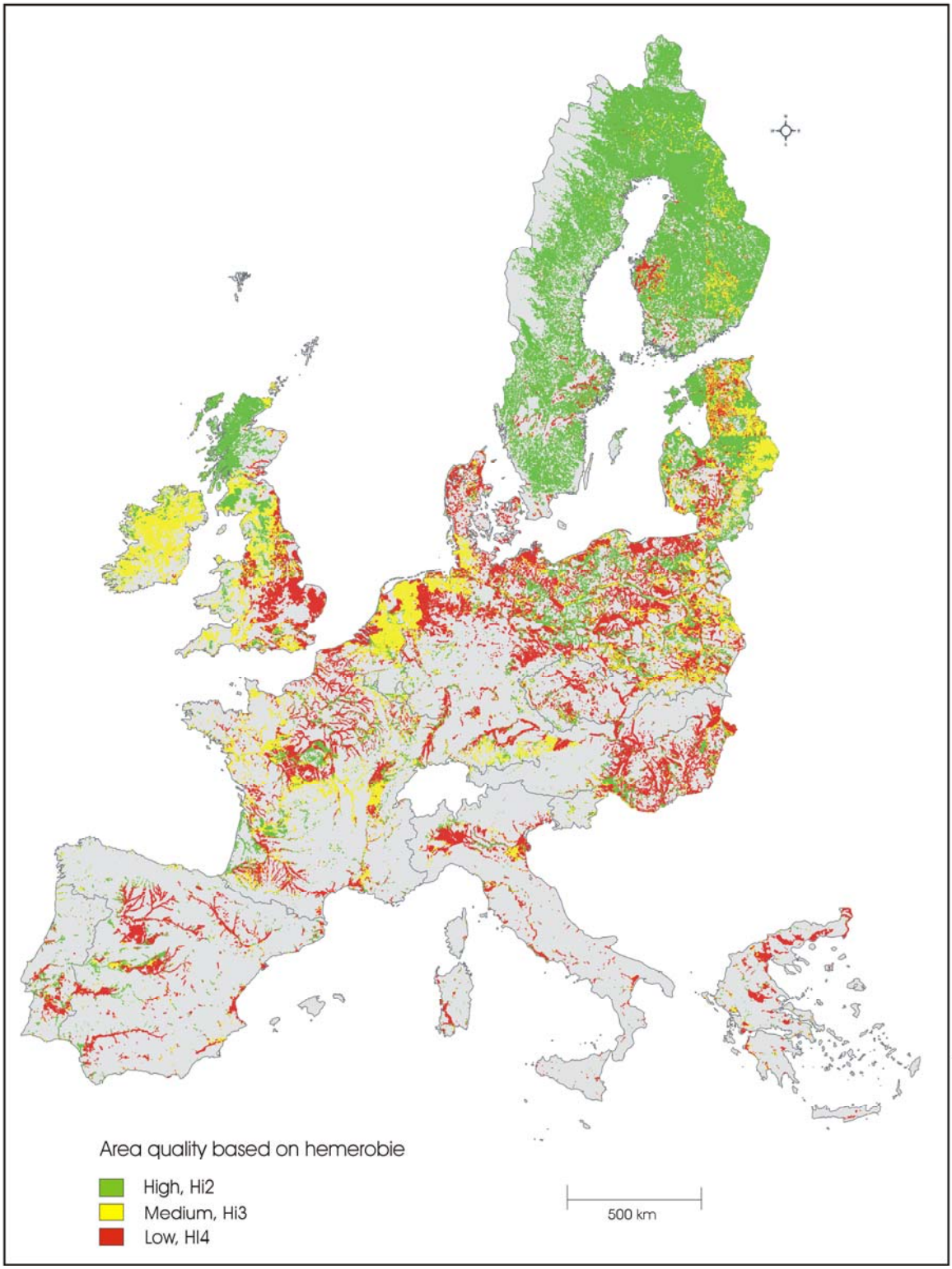


Fig. 8.7 Area quality of potential wetland restoration sites.

Part IV

Synthesis and next steps



Land use change and the impacts of climate change are major threats to biodiversity preservation and the supply of natural resources or services to humanity. This thesis contributes to current discussions of the impacts of climate change and climate change mitigation politics on land use. Especially, analyses of interactions between land use change and the environment are in the centre of this research. It becomes evident that this thesis originates not from a single, well-defined PhD project but rather combines and integrates results of a number of different studies that are only loosely related. Because it did not start off from a clearly defined question or hypothesis this thesis also does not have overall-summarizing conclusions. It is to be considered moreover as "work in progress". Therefore, in this final part the main conclusions from the respective chapters are summarized and some important aspects as well as possible extensions for future research are highlighted.

The studies presented in **part one** prove that it is possible through spatial analysis to determine the areas of Martinique which are highly sensitive to the consequences of coastal hazards (**CHAPTER 1**) and accelerated sea level rise (**CHAPTER 2** and **3**). The results reveal that the expected accelerated sea level rise will accentuate the impacts and broaden the hazard area (**CHAPTER 3**). Coastal squeeze and beach reduction may be the consequences with severe impacts on the island's tourism economy as stated in **CHAPTER 2**. More than 60% of the human coastal resources are at risk under present conditions and this share will increase if sea level continues to rise (**CHAPTER 3**). In the mountainous parts of Martinique the small areas adjacent to the beaches are often the only flat land available and are therefore compactly colonised. A retreat back to the hinterland as adaptation to sea level rise is often very complicated. The development of a Coastal Zone Management Plan considering sea level rise and its impact area as well as elaboration of public information and evacuation plans is therefore of utmost importance. The decision of the optimal adaptation strategy depends on the priorities and financial limitations of the responsible authorities. Public participation in decision-making and resource management shall always be integrated into the planning process. These studies further emphasize that not only low lying islands are exposed to the consequences of accelerated sea level rise, but that also mountainous islands are vulnerable.

The data situation on Martinique is poor and some scientists even concluded that it is widely impossible to conduct vulnerability assessments in the Caribbean because of this lack of adequate data. But nevertheless, it is of importance to address risks and their impact areas also in regions in which the data background is not optimal. Therefore, broad-scale and broad-brush vulnerability analyses are required by many coastal zone managers. As long as adequate data are missing, spatial modelling is a feasible methodology to obtain statements about coastal impacts due to erosion, inundation or sea level rise. However, more background data would also improve the accuracy of the coastal impact and vulnerability assessments. The studies of **part one** show that spatial analysis allows the evaluation of potential coastal hazard and risk areas by using an empirical assessment model. The utilization and interpretation of satellite images and other spatial data can partly compensate missing base data. The methodology is not uncontentious and therefore validation or calibration is highly desirable. In the case of Martinique a validation of the methodology proved to be successful. The methodology of the GIS-based model is easily applicable and allows individual transformation to other islands or coastlines. The publication of the articles presented in **part one** caused (astonishingly) great interest among coastal researchers. One reason might be that it is increasingly recommended to conduct coastal hazard assessments in regions where spatial data are the only data sources. At present the Martinique approach is going to be transformed to be applicable to the Bangladesh coast in a study undertaken by the Department of Environmental Science and Resource Management of the Mawlana Bhashani University in Tangail (Bangladesh), for example.

In the light of the latest hurricane events in the Caribbean region (e.g. Hurricane Noel that caused severe damage on the Caribbean islands) priorities should be given to work on detailed risk area maps for Martinique that also consider climate change impacts on the coastline. This study may help to fill the gap between applied coastal zone management and science. The information gained from the spatial analyses is useful as basis for regional planners of the Martinique coastal zone to conduct more detailed local studies. Therefore, the results and its underlying GIS-based data are opened up to anybody interested in its application. The methodology using spatial analyses is also useful for everybody interested in

determining the present and future vulnerabilities of coastal zones to erosion and inundation if data from hands-on measurements are scarce or not readily available. Further socio-economic aspects can be easily integrated into the model to illustrate human vulnerability. This includes the evaluation of damage costs caused by hurricane or tsunami flooding. The results obtained from the GIS-based model offset the lack of data. The interaction of detailed spatial evaluations and socio-economic models increases the accuracy of the results and will be the main challenge for science. The World Bank, which is going to conduct a study on indicators and GIS-based tools for vulnerability analysis due to sea level rise, already indicated its interest in the Martinique study (J. Kingston, Development Economics Research Group, The World Bank, pers. comm.).

Part two of this thesis deals with the impacts of land use changes on bird populations on the Eiderstedt peninsula in Schleswig-Holstein (Germany). Generally, the landscape here is dominated by extensively used grassland. These grassland areas are home to many bird species, and among naturalists Eiderstedt is considered to be one of the most important bird habitats in Schleswig-Holstein. Ongoing changes in the structure of the regional agriculture towards an intensified cattle production and the growth of biofuels call for a conversion of large shares of grassland to arable farm land. Today, many farmers argue that a distinct expansion of arable farming is the only way to survive economically and that shifts in the overall structure of the regional agriculture necessitate these conversions. Through scenario development the awareness should raised about the potential implications to the environment that might be caused by political decisions (**CHAPTER 4**).

As explained in **CHAPTER 5,** the quality of the Eiderstedt peninsula as breeding habitat for meadowbirds decreases substantially as a consequence of a large scale land use change. As the suitability of the land to serve as breeding habitat declines, the number of individuals supported by the habitats will be reduced at a disproportionately high rate. Even the declaration of the three bird sanctuaries on Eiderstedt is insufficient to counter this trend since the sanctuaries are also negatively influenced by changes in their vicinity, whose influence carries over into the protected areas. Therefore, buffer zones around these bird conservation areas are of paramount importance to preserve the existing habitat quality. The

potential environmental impacts of the land use conversion differ depending on the resulting distribution pattern of agricultural land.

In addition to adverse ornithological impacts, a substantial land use change on Eiderstedt can have an influence on income generated by tourism. Eiderstedt is a famous destination due to its pristine beaches, the Wadden Sea National Park, but also due to extensive grassland areas and the large numbers of breeding and migrating birds to be observed. The general appearance of Eiderstedt to visitors will change if large parts of grassland are replaced by arable farm land for crop production. Subsequent studies will investigate these options more closely. The controversy about land use development can only be solved if a compromise can be reached between the ecological demands of the ornithological fauna, the economic interests of farmers, and the aesthetic expectations of tourists visiting the Eiderstedt peninsula. Public awareness of the benefits accruing from habitat conservation is a necessary but too-often ignored ingredient in achieving cultural sustainability for conservation in human-dominated landscapes (MUHAR 1999; MILLER 2007). Therefore conservationists must invest greater effort in understanding the motivations and perceptions of private landowners and in defining the benefits to landowners by maintaining these habitats. In cooperation with the geography department of the University of Bonn a methodology will be conducted in solving this problem and also to further elaborate on the scenarios as well as to integrate aspects of social sustainability. This study offers not only advanced methodological insights into human-environment interactions but also aims to make the findings applicable to the research area to support possible solutions of the conflict.

Impacts similar to those observed on converted grassland are found in the salt marshes outside the main dikes, where the highest bird densities are generally observed. For these areas, suitable conditions for breeding need to be present in the adjacent hinterland as well if their overall quality as ornithological habitat is to be maintained. The importance of an intact neighbourhood is augmented if the impacts of sea level rise on the salt marshes are considered as well. Because of impossibility of retreat due to anthropogenic infrastructure such as dikes, an accentuated erosion of the unprotected salt marshes due to sea level rise and intensified storm events might take place, leading to the deterioration or loss of

the potentially most valuable breeding areas for grassland birds. The hinterland on the Eiderstedt peninsula could serve as highly suitable substitution habitat, but only if at least current conditions are preserved. A more detailed study about the potential habitat loss due to the erosion of salt marshes and the use of inland grasslands as surrogates is planned. This implies an expansion of the study area at least to the entire German North Sea coast. Additionally, the study will be extended to migratory birds that use the areas as stop-over sites.

Depending on the landscape and objectives, nature conservation management may target single species, groups of species, or whole ecosystems. In **part three** of the thesis we initially focus on freshwater wetlands as biotope complexes. The wetland distribution model SWEDI presented in **CHAPTER 6** estimates wetland sites in the EU-25 countries by distinguishing between existing wetlands and potential restoration sites. The SWEDI data revealed in total five times more wetland potentials than existing wetlands in Europe. The wetland distribution map on European scale has a high spatial resolution of 1 ha for the existing wetlands, respectively 1 km² for the potential sites. Often no clear distinction is made between wetlands and aquatic systems, but in order to regulate and to build a conservation strategy one must first have a clear definition of wetland. The model distinguishes also between different wetland types. This implies a detailed wetland classification. A number of countries have a policy of no-net loss of wetlands that requires mitigation when wetlands are destroyed or degraded (MARSH ET AL. 1996). Often this results in a replacement of diverse wetlands with open-water pools ringed with common marsh plants (NAT RES COUNCIL 2001). The distinction of different wetland types in the wetland model and its integration into the economic optimization model EUFASOM offers the opportunity to promote the establishment of more diverse biotope complexes. An advantage of the potential convertible sites model is its orientation towards physical parameters and the allowance of overlapping wetland types. In this way geographical and physical borders of different wetland types are well reproduced.

Restoration and conservation management are increasingly viewed as complementary activities with restoration often being an important element of conservation management (YOUNG 2000). The GIS-based SWEDI model provides the base data for upscaling the spatially explicit wetland distribution for use in

EUFASOM. As explained in **CHAPTER 7** the connection of the wetland distribution model with the economic land use model EUFASOM investigates the socially optimal balance between alternative wetland uses by integrating biological benefits – in this case wetlands – and economic opportunities – here agriculture and forestry. Several scenarios with different constraints are assessed. This way we can demonstrate the tradeoffs between obtaining higher levels of a conservation target and the increase in cost necessary to obtain it. The results indicate the potential influence of biomass supply on wetland restoration efforts. They further reveal that financial incentives motivate landowners to invest in nature conservation: with rising incentives also increasing wetland area is restored. But, in combination with increasing wetland potentials, food prices will go up as well, because of less food production in favour of wetland restoration efforts. The scenarios also show that protection of existing wetlands is necessary to prevent wetland loss and conversion of wetlands into production land uses. Regional and country-specific differences in wetland potentials exist.

The wetland potentials of the EUFASOM scenarios build the base for the spatial site-selection model (cf. **CHAPTER 8**) that provides an appropriate tool to identify suitable locations for the creation or restoration of wetland systems. Ecosystem restoration is a vital tool in the maintenance and restoration of resilience. But the key question is what the most suitable patterns of revegetated blocks or strips are to achieve an effective habitat network across the landscape (BENNET & REDFORD 2007). In this context, a prioritization framework and the formulation of restoration targets are of importance. CALHOUN (2007) states that "wetland regulations should be designed to conserve an array of wetland functions, not limited to water quality, waterfowl habitat and recreation. They should address cumulative impacts and connectivity of wetland, aquatic and terrestrial resources and be comprehensive enough to protect both individual wetlands and the overall integrity of landscapes in which wetlands occur".

While the site-selection model is only applicable to Germany at present it needs to be expanded for the whole EU in a following step. To work as a spatial decision tool it also needs to be transformed into an applicable user-interface for public utilization. It is also planned to include more countries than the EU-25 states in the model and to extend the application to more ecosystems than wetlands.

Especially in consideration of **part two** it might be of interest to do more spatial investigations about grasslands and their conversion on a European scale, for example.

The spatially explicit analysis of wetland distribution on a European scale is technically challenging, especially when considering its computing memory requirements. In the past, conservationists focused on finer scales only and ignored broader scales. But because the major threats to conservation are land use and land use change, a good deal of conservation action must be directed at the scale of land use through policy, establishing protected areas, ecological restoration, designation of human activities that are compatible with biodiversity protection and the like. It is therefore recommended that the scale of the goals and objectives must match the scale of the challenge (SCOTT & TEAR 2007). Still, there are a lot of opponents against these broad scale analyses. Their main arguments are the uncertainties and simplifications made. Surely, there is still space for improvements not only concerning the base data.

However, knowledge of the extent and distribution of wetlands is important for a variety of applications. The spatial wetland distribution and site-selection model helps to locate sites suitable for renaturation programs, or for the introduction of faunistic corridors respecting the Natura 2000 network of sites. The application of the model in nature conservation issues favours the success in regional conservation planning. Many conservation management strategies focus on individual species only. Emerging landscape-scale processes that affect large numbers of species make these spatially explicit strategies essential (BURGMAN ET AL. 2007). It is therefore planned to integrate the wetland distribution model into an economic wetland biodiversity optimization model. This allows the combination of the ecosystem approach with multi-species assessments to make conservation efforts even more effective.

Another challenge is to assess the wetland function and distribution in the light of global climate change. A current study by SCHNEIDER ET AL. (unpubl.) attempts to quantify the greenhouse gas mitigation potential of European wetlands under utilization of the SWEDI data. In this respect special attention is paid to the heterogeneity of natural conditions, land management, and greenhouse gas fluxes. To account for opportunity cost of wetland protection and restoration, in the study

mitigation potentials are assessed with EUFASOM. In this respect a collaboration with the BMBF promoted project “Klimaschutz - Moornutzungsstrategien” is planned. Further, it is planned to extend the European wetland distribution and site-selection model with hydrogeomorphic parameters so that it can be coupled not only with an ongoing study of global wetland distribution under climate change but also with climate change scenarios themselves.

Concluding, there is a growing interest in applications of GIS solutions in sustainability and global change. GIS provides not only a method to enhance communication between science and the public. Through this it is also possible to evaluate the spatial impacts of political or policy decisions and to contribute to its reformulation.

As the Earth’s population, economies and demands for resources continue to grow, changing land use will pose ever greater challenges to biodiversity. But also the projected climate change and its consequences are going to influence human spatial behaviour. Geography, landscape ecology and other spatial sciences will experience growing relevance in detecting and illustrating these challenges. Only during the last few years, awareness of the relevance of these spatial connections has grown among conservationists and economists. There is an urgent need to identify spatially explicit options for improving the connectivity of fragmented landscapes and to identify and pinpoint human risk and natural hazard areas arising from climate change. Applications of spatially explicit information and analysis tools improve the assessment process. They can also help to identify areas or regions in which the payoff for conservation efforts is likely to be greater, or in which climate change adaptation options are economically feasible and inevitable. There is a need to increase the understanding of how spatial patterns influence ecological or economic processes (WIENS 2007).

The combination of natural sciences with social sciences is a challenging topic. GIS solutions not only build a bridge between social, economic and geosciences but also illustrate the variety of viewpoints and results more clearly for public participation and stakeholder applicability.

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