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Integrating ecological knowledge with legal instruments for nature conservation in river management

Reinier de Nooij

Integrating ecological knowledge with legal instruments for nature conservation in river management

een wetenschappelijke proeve op het gebied
van de Natuurwetenschappen, Wiskunde en Informatica

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Chapter I

General introduction

Nature conservation and valuation in river management

1 Nature conservation in river management

River-floodplain ecosystems belong to the most affected ecosystems in the world (Calow & Petts, 1992, 1994; Tockner & Stanford, 2002) due to human preference for river systems as a habitat. Floodplains are defined as "areas of low-lying land that are subject to inundation by lateral overflow water from rivers or lakes with which they are associated" (Junk & Welcomme, 1990). Riverine floodplains are among the earth's most distinctive landscape features, and are of great cultural and economic importance. Throughout history, many civilizations arose in fertile floodplains, cultivating and using their rich natural resources. Therefore, riverine floodplains have served as nuclei of urban development and exploitation of the natural functions of river systems (Tockner & Stanford, 2002). The intimate relationship between development of societies and river floodplain ecosystems has resulted in very complex and diverse systems all over the world. However, human influence has, notably in Western Europe, also resulted in severe impoverishment of biological diversity in river systems (Petts, 1989).

1.1 River systems and biodiversity

The riverine landscape is characterised by a diverse array of landscape elements, including surface waters (a gradient of flowing and stagnant water bodies), the alluvial aquifers, riparian systems (alluvial forests, marshes, meadows) and geomorphic features (bars and islands, ridges and shallow channels, levees and terraces, fans and deltas, fringing floodplains, wood debris deposits and channel networks (Ward ***et al.***, 2002). The Rhine-Meuse delta is also highly esteemed for its cultural-historical landscape qualities such as winding dikes, scour holes and cropped willows (Haartsen ***et al.***, 1989). This landscape diversity not only constitutes recreational and aesthetic values: river floodplain ecosystems are characterized by high biodiversity and productivity.

The Convention on Biological Diversity, which resulted from the 1992 Rio 'Earth Summit', defines the conservation of biodiversity as its main objective. According to the Convention, biodiversity is defined as "the variability among living organisms (...) including (...) ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems" (UNEP, 1992). This definition distinguishes three levels (genetic, species and ecosystem level). Noss (1990) looks at biodiversity from a different angle and distinguishes three components: compositional, structural and functional biodiversity. Each of the three levels of diversity can be characterised and described using these three components (Le Maitre & Gelderblom, 1998). Composition then refers to presence and abundance of flora and fauna species and populations, types of ecosystems in the area and local genetic varieties. Structure describes how the elements of biodiversity, including genes, species, habitats, geomorphic patterns and cyclic phenomena are organised in space and time (i.e. spatial and temporal patterns). Functional diversity refers to physical, biological or bio-physical processes structuring ecosystems and communities, such as disturbance regimes, succession, population dynamics and gene flow (Ward ***et al.***, 1999). Composition, structure and functioning are interdependent and interwoven aspects of ecosystems (Noss, 1990).

Fluvial dynamics (inundation, erosion, transport, deposition) are the most important processes controlling the structure and development of the riverine landscape and also constitute the natural disturbance regime primarily responsible for sustaining a high level of structural and functional diversity (Figure 1). This diversity can be expressed using the concept of ecotopes. Ecotopes are defined as spatial units of a certain extent, which are relatively homogeneous in terms of vegetation structure, succession stage and the main abiotic site factors that are relevant to plant growth (Klijn & Udo de Haes, 1994). In the Dutch River Ecotope System, river ecotopes are identified on the basis of hydrodynamics, morphodynamics, management dynamics and land use (Van der Molen ***et al.***, 2003). This definition includes anthropogenic dynamics, and therefore recognises the importance of human activity for the composition, structure and functioning of the riverine landscape.

The riverine landscape itself can be characterised as the transitional zone between aquatic and terrestrial environments (Junk, 1989). Therefore it is by its very nature an ecological gradient. Although individual landscape features (ecotopes) may exhibit high turnover, largely as a function of the interactions between fluvial dynamics and successional phenomena, their relative abundance in the river corridor tends to remain constant over ecological time (Ward ***et al.***, 2002; Geerling ***et al.***, 2006).

River ecosystems are species-rich as a result of the high spatio-temporal heterogeneity of physical habitat, created by the dynamic interactions of water and land (Figure 1). The dynamic interaction between water and land is the principal process that produced river-floodplains, maintains them, and has affected the adaptations of biota that have evolved therein (Bayley, 1995; Junk, 1989; Van den Brink, 1994, 1996; Ward & Tockner, 2001; Aarts ***et al.***, 2004). Hydromorphodynamic processes shape the template on which evolution forges characteristic life-history strategies in river systems (cf. Southwood, 1977). The Intermediate Disturbance hypothesis (Huston, 1979) predicts that biodiversity is highest with intermediate levels of fluvial dynamics, and maximum levels of spatial heterogeneity.

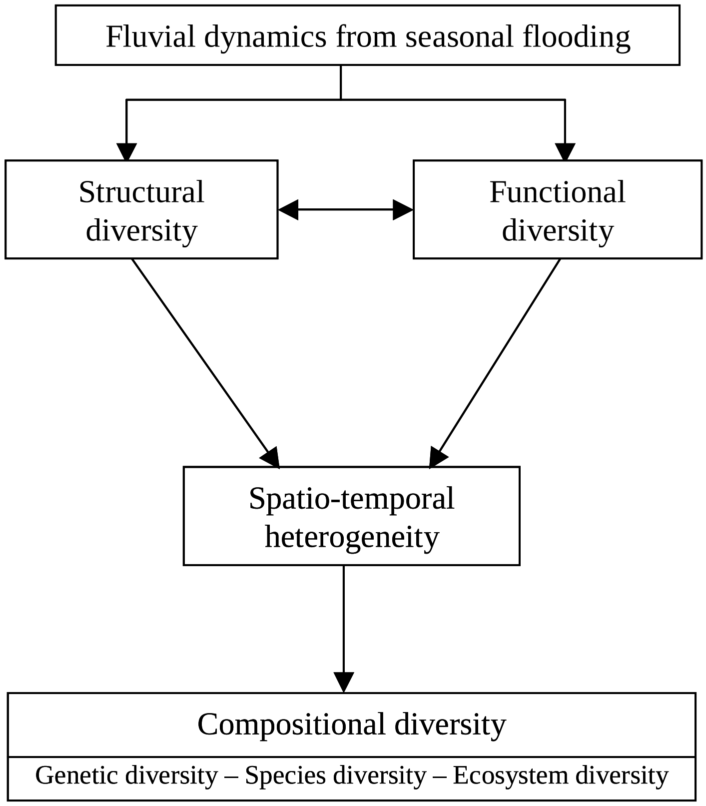

Figure 1. Processes and mechanisms leading to high biodiversity in river-floodplain ecosystems (modified from Ward ***et al.***, 1999).

1.2 Human influence on riverine biodiversity in north-western Europe

Human influence throughout the centuries has had positive as well as negative effects on biodiversity of river systems in north-western Europe. There is evidence that already around 8000 BC humans locally cleared the forested natural levees and floodplains in the Rhine basin and put them to use for hunting purposes (Bos & Urz, 2003). In the Netherlands, agricultural occupation of natural levees and river dunes already occurred in the middle of the fourth millennium BC (Groenman-van Wateringen, 1978). In this initial period of human influence on the structure and functioning of river systems, landscape diversity increased, and so did probably species diversity. Gradually, however, the impoverishing influence became more and more dominant. Humans have been modifying river systems in numerous ways, which led to severe reduction of spatial and temporal heterogeneity (Figure 2). Many landscape elements created by the dynamic interplay of water, land and biota have been replaced by man-made structures and man-dominated ecosystems.

In the Netherlands for example, the gradual construction of high water free dwelling zones and dikes in the basin of the river Rhine resulted in a closed dike system around 1400 AD (Harten, 2000), which restricted fluvial dynamics to the narrow parts of the alluvial system between the dikes. This drastically decreased dynamics in the hinterland, but strongly increased dynamics within the river channel (Lenders, 2003). The reduction of intermediate dynamics had a devastating effect on biodiversity in river-floodplain ecosystems. Similar processes occurred in the river basins of all north-western Europe (Nienhuis ***et al.***, 1998; Havinga & Smits, 2000).

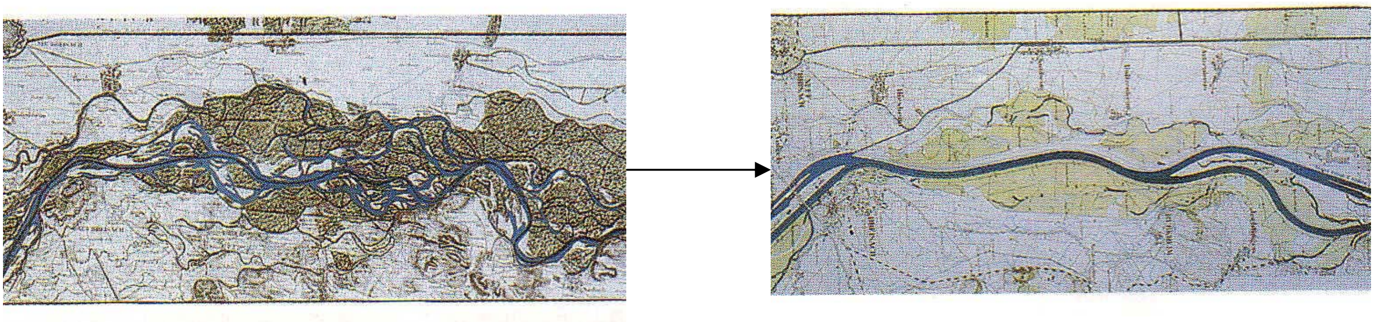

Figure 2. Changes in the riverine landscape due to physical normalisation and land-use change in an upper reach of the river Rhine, just below Strassbourg, in 1828 and in 1963 (modified after Cioc, 2002).

Physical normalisation of the river Rhine by cutting off meanders (18th – 20th century), construction of sluice-dams and fixation of the summerbed by construction of groins and river bank reinforcements, mainly aimed at increasing opportunities for shipping and safety against flooding, further diminished intermediate dynamics (Havinga & Smits, 2000). Apart from intensified agricultural land use, sand and gravel extractions in the 20th century had massive impacts on the structural diversity of river floodplain ecosystems. The combined effects of physical normalisation and land use change on the river landscape are visualised in Figure 2. The decrease of structural diversity and functional (process) diversity led, in combination with the reduction of connectivity, to severe biodiversity decline (Figure 3). In terms of landscape ecological units, especially alluvial forests, natural levee pastures, marshy floodplain pastures and side channels have almost disappeared from the landscape (Middelkoop ***et al.***, 2005). Furthermore, water pollution and the facilitation of invasive species have had profound impoverishing impacts on the diversity of the (native) species composition of river systems (Petts ***et al.***, 1989; Cals ***et al.***, 1998; Nienhuis ***et al.*** 1998; Smits ***et al.***, 2000; Grift, 2001; Bij de Vaate ***et al.***, 2002). Figure 3 shows how all these factors combined have led to severe impoverishment of biodiversity in river-floodplain ecosystems in north-western Europe (Petts, 1989).

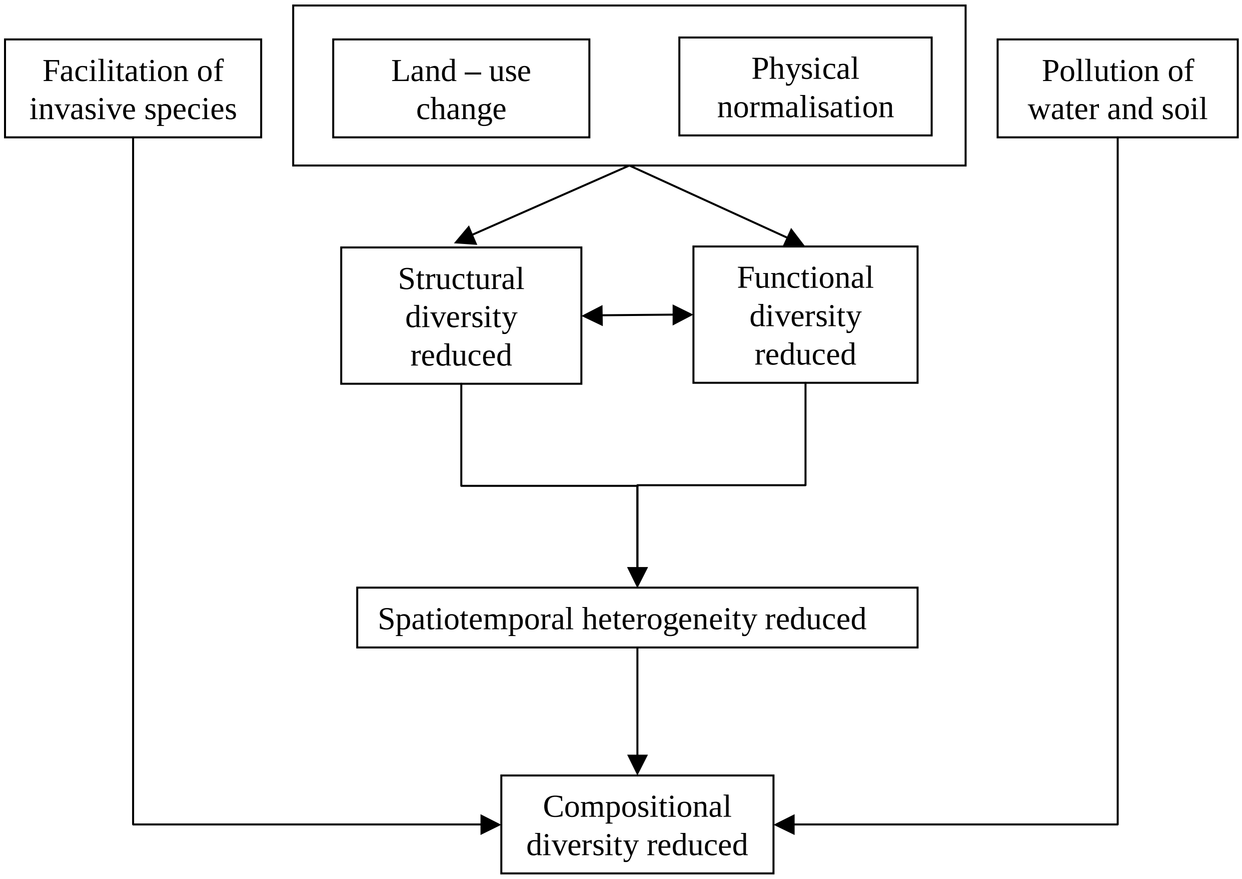

Figure 3. Human influence and mechanisms leading to the decline of biodiversity in river-floodplain ecosystems (modified from Ward, 1998).

In the last decades, efforts are made to reverse this trend. The efforts include improvement of water quality, and rehabilitation of patterns and processes that are natural to river-floodplain ecosystems. Rehabilitation measures include removal of summer dikes, and a change from agricultural management to a strategy that includes influence of river dynamics and low-density grazing by horses and cattle. These measures increase the surface area of ecotopes like natural levee pastures, river dunes, alluvial forests etc., which became rare in the past. In the Netherlands, the effects of these measures are promising but limited (Nienhuis ***et al.***, 2002). Strong boundary conditions for navigation and safety and the small scale of the measures still put heavy constraints on rehabilitation. Van der Molen & Buijse (2005) conclude that, although rehabilitation processes are locally successful, the various projects did not yet significantly contribute to ecological recovery of the river on a higher spatial scale.

Recent flood risk reduction plans, such as the ***Rhine Action Plan on Flood Defence*** (ICPR, 1998)***,*** the ***Meuse High Water Action Plan*** (ICPM, 1998), the ***Declaration of Arles*** (Anonymous, 1995) and the Dutch policy plan ***Space for Rivers*** (Dutch Ministry of Transport, Public Works and Water Management, 1996; 2005), aim at improved water management and flood risk reduction through spatial planning. The goal of these plans is protection of people

and goods against flooding (safety) while integrating ecological improvement of the rivers Rhine and Meuse and their floodplains.
Large-scale reconstruction measures are being prepared and implemented in river basins of north-western Europe for the purpose of flood risk reduction, ecological rehabilitation and infrastructural improvements (Van Stokkom ***et al.***, 2005). These measures will have far-reaching consequences for the physical structure and dynamics, and hence for the ecological functioning, of river-floodplain ecosystems (Nienhuis ***et al.***, 1998; Smits ***et al.***, 2000).

In the Netherlands, the plans include measures such as large scale floodplain excavation, reopening of secondary channels, river dike repositioning, removal of elevated areas and riverbed lowering (Van Stokkom ***et al.***, 2005). The floodplains of the rivers Rhine and Meuse in the Netherlands will undergo yet other significant changes in their ecotope distributions. In general, a shift will occur from agricultural ecotopes to more natural ecotopes like extensively used grassland with herbaceous vegetation, natural levee pastures and floodplain marshlands.

Although the measures can have positive effects, serious problems with nature conservation policy and legislation can arise. There is a tension between the goals of rehabilitation, nature conservation and flood risk reduction (Nienhuis & Leuven, 2001; Nienhuis ***et al.***, 2002). Rehabilitation refers to the reintroduction of processes in the river floodplain ecosystem, in order to improve the structural, functional and compositional diversity. Conservation means safeguarding present values for the future. Current strategies for flood risk reduction aim at giving more room to river dynamics in river floodplains. Rehabilitation and reconstruction measures will not only have positive effects, but may also lead to the disappearance of species protected by national and/or international legislation.

1.3 Nature conservation policy and legislation

Conservation of biodiversity is one of the key issues of world-wide environmental policy. Nature conservation policy in the EU aims at ecological rehabilitation and conservation of biodiversity. Nature conservation legislation is an important instrument for governments to carry out and provide a legal basis for this policy. Conversely, policy may be developed to ensure that management practice is consistent with the law. Policy is formed by covenants, administrative goals and instruments for daily practice. Law consists of legislation, jurisprudence, conventions and treaties. In general, nature conservation policy and legislation are being developed in close interaction.

On the international level, nature conservation legislation is provided by the Conventions of Bern, Bonn and Ramsar, and the Convention on Biological Diversity. The legislative framework for nature protection in Europe consists of the Birds Directive (Council Directive 79/409/EEC) and the Habitats Directive (Council Directive 92/43/EEC). In the European Union, the Birds Directive and the Habitats Directive are considered to cover the mentioned conventions. However, there is some discussion whether the Bern Convention may be completely disregarded (Maes & Neumann, 2002). The purpose of the Directives is to maintain or restore Europe's wildlife and their habitats at a favourable conservation status in their natural range, by means of designating protected areas and protected species. Significant negative impacts of human activities on areas and species protected by these directives are not allowed, unless (i) there are no alternative solutions and (ii) there are imperative reasons of overriding public interest that demand these activities. Even if these two conditions have been met, the negative impacts on protected areas have to be compensated for. River managers are therefore obliged to take protected areas and species into account in planning and effect assessments concerning physical reconstruction and management (e.g. Environmental Impact Assessments and Strategic Environmental Assessments). The Birds and Habitats Directive both have regulations for species protection as well as area protection. In the Netherlands, the species protection component of the Birds Directive and Habitats Directive is implemented by means of the Flora and Fauna Act; the area protection component into the Nature Protection Act 1998.

The European Water Framework Directive (WFD, Council Directive 2000/60/EC) provides legislation for the management of water systems in the EU. According to the WFD, a good ecological status for natural rivers and a good ecological potential for heavily modified waters have to be achieved by 2015. The WFD is aimed at providing common principles in order to ensure Member States efforts to improve protection of Community water to protect aquatic ecosystems, and terrestrial ecosystems directly dependent on them. This directive clearly indicates that species and habitats mentioned in the Birds Directive and Habitats Directive must be taken into account in policy and management, and also requires the selection of species for specific water types.

A distinction can be made in hard law and soft law. Hard law is legally binding (i.e. offers the opportunity for sanctions) and can be directly called upon. Soft law is not legally binding and can not be directly called upon. However, it represents consensus about elementary topics and often forms the basis for newly developed hard law (Backes ***et al.***, 2006). The Birds Directive, Habitats Directive and Water Framework Directive are clear examples of hard law; the status of the Conventions of Bern, Bonn and Ramsar is less clear but is generally regarded as being soft law. The Convention on Biological Diversity is without doubt an example of soft law. Policy by itself does not have possibilities for sanctions. However, as mentioned above, policy is often grounded in, and made legally binding by legislation. Policy gives directions in everyday practice and is therefore highly relevant for management and physical planning. In the development of environmental policy, one can distinguish more or less regular patterns which have grown into so-called policy principles (Van Geest & Hödl, 1998). These policy principles can be seen as having an intermediate character between legislation and policy. They give protocols which are important in practice, and can be called upon, but are not legally binding. An example of policy principles within the context of nature conservation is the attention paid to Red Lists.

Red Lists play an important role in prioritisation of conservation efforts. The IUCN Red List methodology (IUCN, 1993; 1994) classifies species into seven categories on the basis of scientific data on abundance and trends of species populations. These categories represent levels of extinction risks. The obligation to compile Red Lists comes from the convention of Bern. In 2004, a Ministerial Decree made the Red Lists official in the Netherlands. Red Lists therefore have a legal status, but species on Red Lists are not automatically legally protected. Red Lists are important policy and management instruments because they are readily used in day to day practice and have a strong signal function. Furthermore, the Red Lists form the basis for the selection of protected species, as is the case with Birds Directive, Habitats Directive and the Bern and Bonn Conventions. Red listed species are frequently used in addition to protected species in environmental impact assessments (Slootweg & Kolhoff, 2003). The European directives and international conventions, national legislation, and policy principles like the importance of Red Lists make protected and endangered species an important issue in river management. Together, they form the political-legal framework, which gives the instruments and regulations for conservation of biodiversity, and the requirements for input of ecological information in planning processes and legal procedures.

2 Multiple approaches towards biodiversity valuation

Biodiversity is a concept that is context dependent and the way it is used always reflects a way of thinking (Mayer, 2006). There are various approaches for using the concept ‘biodiversity’ and giving meaning to the term. Different approaches focus on different dimensions of biodiversity and have their own methods for valuation. In this section the broad nature of the biodiversity concept is briefly highlighted. Two approaches towards biodiversity, i.e. ecological and political-legal, are discussed in more detail.

2.1 Approaches towards biodiversity

Within the context of the Convention on Biological Diversity (UNEP, 1992), biodiversity has many dimensions (e.g. ecological, social, economical and political; Groombridge, 1992; Putterman, 1994; Olembo, 1995; Orlove & Brush, 1996; Pearce ***et al.***, 1996; Pimentel ***et al.***, 1997; Swanson, 1997; Edwards & Abivardi, 1998). Biodiversity represents a broad and integrated perspective (Huston, 1994; Schulze & Mooney, 1994; Rosenzweig, 1995) and a heightened concern for threats to gene pools, species and habitats on a global scale (Wilson, 1992; Ricklefs & Schluter, 1993; Mooney ***et al.***, 1996). In this thesis two dimensions play a role: the ecological and the political-legal dimension, corresponding to an ecological-scientific and a political-legal approach.

Ecological science aims at objective inquiry into patterns and processes in the biosphere and the interactions between organisms and their environments as causes and mechanisms leading to biodiversity. Within a scientific context, biodiversity is a synthetic concept, inextricably linked to various ecological concepts such as succession, patch dynamics and connectivity. The concept draws upon disciplines such as (landscape) ecology, biogeography, (population) genetics and evolutionary sciences (Ward ***et al.***, 1999).

Political-legal considerations are involved in conservation and restoration of biodiversity, based on the notion that biodiversity represents ecological, economical and social value, and on the ethical argument of stewardship (Council Directive 92/43/EEC; UNEP, 1992). The political-legal framework gives guidelines and general principles that must guide human activity. The objective of the Convention on Biological Diversity (UNEP, 1992) is “the conservation of biological diversity, the sustainable use of its components and the fair and equitable sharing of benefits arising out of the utilization of genetic resources, including appropriate access to genetic resources and by appropriate transfer of relevant technologies, taking into account all rights over those resources and to technologies, and by appropriate funding”. The Model Act on the Protection of the Environment (Council of Europe, 1994) states that “any action shall avoid having any substantial adverse effect on biodiversity”. The World Conservation Strategy (IUCN ***et al.***, 1980) calls the conservation of genetic diversity one of the three main objectives of living resource conservation. The World Charter for Nature states as a general principle: “the genetic viability on the earth shall not be compromised; the population levels of all life forms, wild and domesticated, must be at least sufficient for their survival, and to this end necessary habitat shall be safeguarded”. In any case, conservation of biodiversity is an activity that results from the assignment of values by humans, whether intrinsic or functional, to species, ecosystems or diversity itself.

2.2 Biodiversity valuation

Biodiversity is a framework concept, referring to the variety of life on earth. Biodiversity in this sense is not measurable or quantifiable. However, specific features of biodiversity, e.g. species richness of vascular plants, can be quantified. When mentioning biodiversity, reference must always be made to the approach chosen, and to the feature of biodiversity concerned (Mayer, 2006). Because the issue of biodiversity plays a role on many different spatio-temporal scales and biological levels (e.g. genes, species, higher taxonomic levels, ecosystems), also the level and scale must be made explicit. For nature conservation purposes, the broad definition of biodiversity leads to problems in making this concept operational in every day practice, especially at the intra-species and ecosystem levels (Lenders ***et al.***, 1998). Therefore, in practice the concept is mainly expressed in terms of inter-species biodiversity (i.e. diversity of species).

Ecological valuation

Each of the three levels of diversity (i.e., genetic, species, and ecosystem level) can be characterised and described using three components of biodiversity: composition, structure and process (see paragraph 1.1). Composition is measured by presence and abundance of entities such as flora and fauna, types of ecosystems in the area and local varieties (genetic). For this purpose, traditional biological approaches, based on species abundance or species richness, are frequently used (see for some examples on river and floodplain ecosystems: Schnitzler, 1994; Van den Brink ***et al.***, 1994; Obrdlik ***et al.***, 1995; Buijse & Vriese, 1996; Grévilliot & Muller, 1996; Van den Brink ***et al.***, 1996). Biological indices (e.g., the Shannon index, Margalef's diversity index and Menhinick's index; Magurran, 1988) use detailed information on species richness, numbers of individuals per species, taxonomic group or habitat (abundance), rarity, completeness or naturalness of an ecosystem or an area (Magurran, 1988).

Another approach for ecological valuation of composition is selection of focal species (Brooker, 2002). These are species with a relatively high degree of extinction risk and/or an important ecological function. According to Simberloff (1998), species with a large influence are called key-stone species. Species can also be indicative for a certain environmental quality or characteristic of a certain type of ecosystem (indicator species). Moreover, species can have a so-called umbrella function. Their habitat requirements are assumed to cover those of a lot of other species. Protection of umbrella species may therefore result in the protection of many others. However, ecological scientists still heavily debate the umbrella species and key stone species concepts as well as the identification of these types of species (Lindenmayer & Fischer, 2003).

Structure of biodiversity is quantified using heterogeneity of landscapes and ecosystems, incorporating surface distribution and the numbers of ecotopes in an area, geomorphic patterns, gradients microhabitat structure and structural aspects of genetic diversity. Process diversity of landscape and ecosystems can also be quantified, using the features disturbance, connectivity, energy flow, population dynamics, gene flow etc. (Ward ***et al.***, 1999).

Political-legal valuation

Policy and legislation based biodiversity indicators should be regarded as practical tools for estimating human impact on biodiversity, with emphasis on goals concerning biodiversity. These indicators can be used for impact assessment and for measuring progress towards meeting legal obligations.

Policy and legislation selects from the composition of ecosystems a number of species, which are considered the most relevant. These species are marked as protected (legislation) or target (policy). The weights given to species reflect their legal status. For example, species protected by the Habitats Directive (hard law) are considered more important than species mentioned in the Convention of Bonn (soft law). Concerning structure and process, specific features of protected areas can have a protected status. In the Habitats Directive for example this is described by means of the term “integrity of the site concerned” (Art. 6) and “appropriate features of the landscape which are of major importance for the wild flora and fauna” (Art. 3).

In the practice of management and physical planning, ecotopes are used in order to describe structure in space and time. Ecotopes can also represent specific processes, such as hydrodynamics and morphodynamics in river systems (paragraph 1.1). In a political-legal approach towards biodiversity, a simplification occurs with respect to composition, structure and process. However, this simplification is necessary for definition of feasible goals and required actions.

3 Scope, goals, questions and outline of this thesis

3.1 Scope

This thesis concerns integration of ecological knowledge with policy and legal instruments for nature conservation in river management. The political-legal and ecological implications of this integration within the context of river management are studied. Within river management, I focus on physical reconstruction and management aimed at flood risk reduction (safety) and ecological rehabilitation. The study area concerns the rivers Rhine and Meuse in the Netherlands, Germany, Belgium and France. Concerning biodiversity, the emphasis is on species and taxonomic groups, spatial scales relevant for floodplain landscapes and temporal scales up to 50 years. Political-legal aspects of nature conservation concern international conventions (Conventions of Bern and Bonn), EU legislation in the form of the Habitats Directive and Birds Directive and national legislation (Flora and Fauna Act), and the Red Lists. The Water Framework Directive is not included in this thesis, because implementation and in particular the species selection process was not yet complete during the course of our research.

The central idea is that a model is required for integrating ecological knowledge and information with political and legal considerations concerning biodiversity. This model will facilitate the input of ecological information into landscape ecological studies, planning and decision making processes, and legal procedures. River management can then be more efficient and consistent, optimising the balance between safety and nature conservation.

3.2 The need for integration of ecological and political-legal perspectives

Ecological and political-legal perspectives must be related and combined (Herzog, 2000; Thanasis, 2000), otherwise:

1. Policy makers may develop indicators with very limited ecological meaning.
2. Ecologists gather knowledge and information that is difficult to use within a political-legal context.

Within the necessary limitations that arise from feasibility considerations, scientific methods can be applied to insure that policy is based on scientific insights. Models can then be built that organise knowledge on species and ecosystems and information concerning presence of species and ecotopes in a way that is relevant to decision making. Moreover, this can show negative consequences of limitations and missing essential elements in the political-legal system for nature conservation, and can help improving legislation. This combination and mutual adaptation of ecological knowledge and legal instruments is the definition of integration used in this thesis.

Concerning river management, valuation of the effects of physical reconstruction measures is necessary in order to judge these measures on their pros and cons as far as their impact on biodiversity is concerned. Therefore, in planning such measures, it is important to take into account the present and potential value of river related ecotopes. On the one hand, flood reduction measures offer opportunities to recover lost ecotopes by giving literally more space to rivers, thus allowing natural processes to take place again. On the other hand, flood risk reduction measures can seriously endanger present natural values. The outcome largely depends on the degree of effort put into tailor-made designs that optimise the balance between flood risk reduction and improving the ecological status of the river and its floodplains.

Moreover, European jurisdiction on Environmental Impact Assessment procedures and physical planning has shown that it is compulsory to take into account the conservation policy and/or legal status of areas and species in the process of decision-making. If legislation concerning species protection is neglected, this can result in serious delay or even prohibition of implementation of planned measures.

3.3 Goal

The goal of this thesis is to design, apply and evaluate a scientifically underpinned model for integration of ecological knowledge and information with legal instruments for nature conservation in river management, and to show possibilities and limitations of application of this model in evaluation studies and impact assessments concerning river management measures. Furthermore, the implications of the application of the model for management and physical planning will be highlighted.

3.4 Research questions

The following seven research questions are derived from the abovementioned goal:

1. How can a transnational model (BIO-SAFE) be developed for evaluation of and impact assessment regarding biodiversity of river-floodplain ecosystems, integrating ecological knowledge with legal and policy instruments for nature conservation in river management?
2. To what extent does this model yield information indicative for or complementary to a conventional biodiversity quantification method?
3. How valid and sensitive is assessment of impacts of river-floodplain reconstruction using this model?
4. What are the consequences of river floodplain reconstruction and management measures for protected and endangered biodiversity?
5. What are the implications of application of BIO-SAFE in evaluation studies, impact assessment and ecotope valuation for optimisation of river management?
6. What are the possibilities and limitations for integration of ecological knowledge with legal instruments for nature conservation in river management?
7. What is the contribution of the results presented in this thesis to integration in river management?

3.5 Outline

Figure 4 shows the outline of this thesis. The theoretical background, concerning nature conservation, flood risk reduction and biodiversity valuation, is described in this introductory chapter. The problem definition concerns the relationship between floodplain reconstruction and management, legal instruments for nature conservation and ecological knowledge and information.

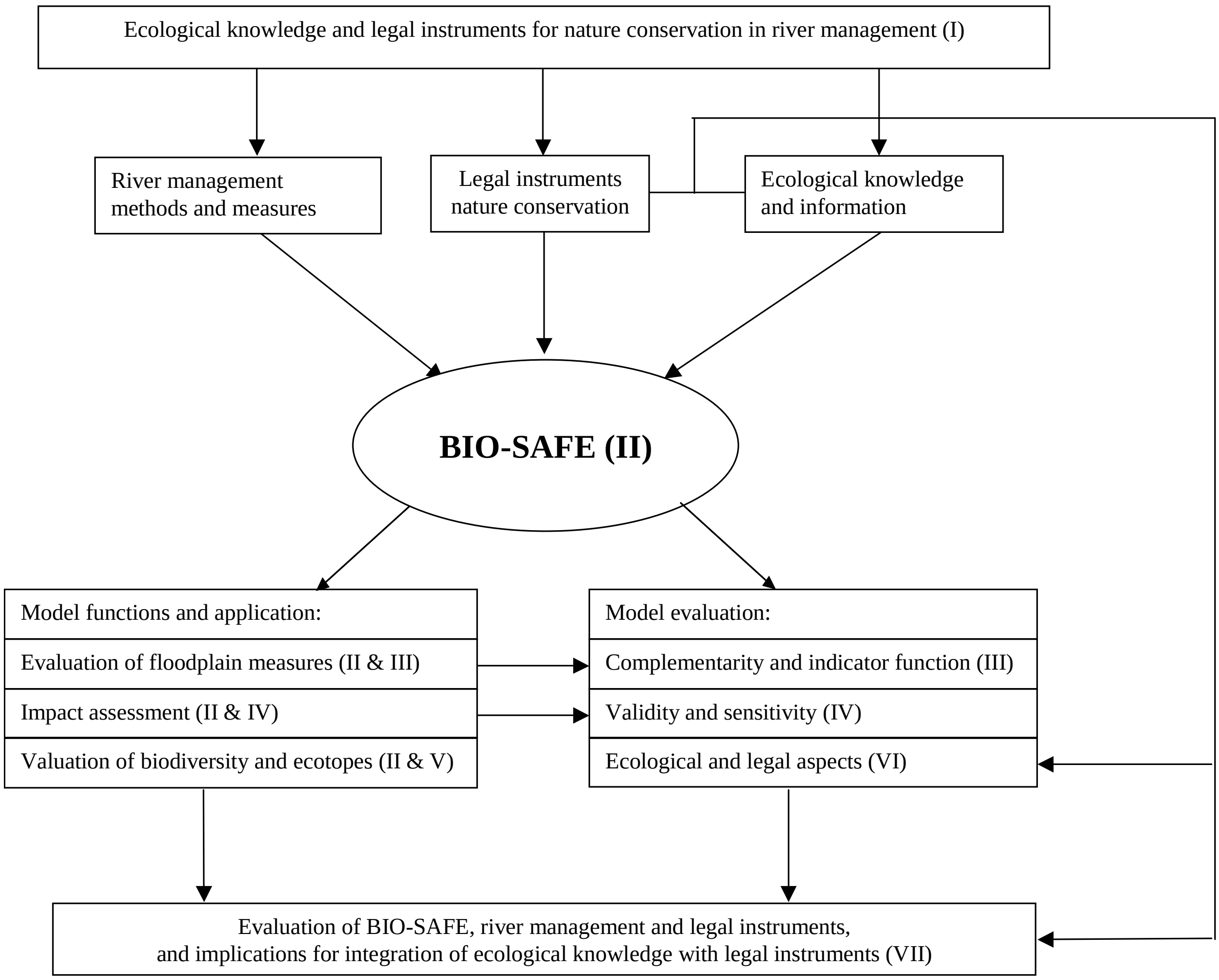

Figure 4. Coherence and outline of this thesis. The Roman numbers refer to the chapters in this thesis.

Chapter II describes the development of an operational and transnational model for evaluation of and impact assessment for river-floodplain ecosystems based on protected and endangered species (BIO-SAFE; research question 1). BIO-SAFE is the acronym for Spreadsheet Application For Evaluation of BIOdiversity. Its possible applications are shown by means of various case studies throughout the chapters of this thesis.

Chapter III focuses on the possibilities for evaluation of rehabilitation measures in floodplains and the complementary information yielded by BIO-SAFE, and the indicator function of the model (research question 2).

The application of the model in scenario analysis and impact assessment is studied in chapter IV. This chapter investigates the validity and sensitivity to value assignment of BIO-

SAFE (research question 3). The results of model application in chapter II, III and IV are used to draw conclusions concerning the consequences of current river floodplain reconstruction and management strategies for protected and endangered biodiversity (research question 4).

In chapter V BIO-SAFE is applied for valuation of ecotopes and hydrodynamic conditions in order to give recommendations concerning the realisation of nature conservation goals (research question 5).

In chapter VI the relation between ecological theory and practice and legal instruments for nature conservation is analysed and linked to river management practice (research question 6). This analysis is used in the synthesis to evaluate BIO-SAFE with respect to its integration of ecological and legal concepts and approaches concerning biodiversity and species traits (research question 7).

In chapter VII (the synthesis) the possibilities and limitations of the model are summarized and recommendations for improvement are given. Furthermore, the implications of nature conservation legislation for reconstruction and management and vice versa, as well as the opportunities for integration of ecological knowledge with legal instruments for nature conservation in river management are discussed.

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CHAPTER II

Construction and application of BIO-SAFE

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Abstract

Assessing actual and potential biodiversity of river-floodplain ecosystems on the basis of policy and legislation concerning endangered and protected species is necessary for consistency between different policy goals. It is thus a prerequisite to sustainable and integrated river management. This paper presents BIO-SAFE, a transnational model that quantifies the relevance of species and ecotopes, characteristic of the main channels and floodplains of the rivers Rhine and Meuse, on the basis of international treaties and directives and national Red Data Lists. BIO-SAFE was developed into a tool for biodiversity assessment with regard to design and evaluation of physical planning projects, Environmental Impact Assessments and comparative landscape-ecological studies. It was conceived to be applicable in Germany, France, Belgium and the Netherlands.

Taxonomic groups involved are higher plants, birds, herpetofauna, mammals, fish, butterflies and dragon- and damselflies. The linkage of habitat requirements of species to ecotopes allows the user to derive information at the level of several ecotope types and scales. The model requires input data on presence of species and/or surface area of ecotopes. BIO-SAFE has been applied to flood risk reduction projects along the rivers Rhine and Meuse. Results show that BIO-SAFE yields quantitative information regarding the degree to which actual situations, reconstruction designs and developments of species and ecotope composition meet national and international agreements on biodiversity conservation.

Attuning biodiversity conservation and flood risk reduction measures is a major issue in applied ecology and spatial planning. Assessments with BIO-SAFE can help find an optimal balance. Because of its policy-based character, BIO-SAFE yields information that is complementary to ecological biodiversity indices, single-species habitat models and ecological network analysis. The development of BIO-SAFE was based on species characteristic of rivers and floodplains, but the method can easily be applied to other ecosystems as well.

1 Introduction

Two major goals of sustainable river management of large rivers in Europe are the reduction of flooding risks, i.e. defined as a combination of probability and consequences of adverse events, and the conservation and rehabilitation of biological diversity. In the coming decades the physical structure of river basins of north-western Europe will undergo significant changes as a result of large-scale reconstruction measures that are currently planned. These measures include lowering of the riverbed and floodplains, removal of raised areas, river dike diversion and construction of retention basins. The measures aim at increasing the water retaining capacity upstream and water storing capacity downstream to prevent future damage from flooding while integrating ecological improvement, and to support economical development by improvement of navigation and creating new infrastructure (Nienhuis ***et al.***, 1998, 2002; Smits ***et al.***, 2000). The measures will have far-reaching impacts on several functions and characteristics of river basins, among which is biodiversity. Flood defence measures can offer opportunities to increase the biological diversity, but they can also seriously endanger present natural values and biodiversity potentials of river ecosystems. Conservation of biodiversity is one of the key issues of world wide environmental policy. According to the Convention on Biological Diversity, which resulted from the 1992 Rio “Earth Summit”, biodiversity is defined as “the variability among living organisms (...) including (...) ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems". As emphasised by Noss (1990) and Wilson (1992), conservation of biodiversity involves more than just species diversity or endangered species.

Natural river ecosystems are species rich as a result of the wide variety of habitat in space and time, created by the dynamic interactions of water and land. The dynamic interaction between water and land is the principal process that produced river floodplains, maintains them, and has affected the adaptations of biota that have evolved therein (Bayley, 1995; Junk ***et al.***, 1989). Biodiversity in river ecosystems can thus be used as an indicator of ecological improvement. Instruments are needed to evaluate the effects of river engineering measures on biodiversity. These can be based on the ecological network function of river ecosystems (***e.g.*** Foppen & Reijnen, 1998; Jongman, 1998; Lenders ***et al***, 1998b; Leuven ***et al.***, 2002) and on the biodiversity potential of landscape ecological units. In these two complementary approaches, riverine landscape ecological units are key elements. The second approach gives the opportunity to take large numbers of species into account, and to include the policy status of endangered or protected flora and fauna (Lenders ***et al.***, 2001). In the course of time, several standardised biological methods to express inter-species biodiversity were developed and used for conservation purposes (***e.g.***, the Shannon index, Margalef’s diversity index and Menhinick’s index; Magurran, 1988). Recently, the need for policy and legislation based biodiversity indicators in addition to biological indicators is acknowledged (Thanasis, 2000; Watt ***et al.***, 2000). Models that integrate these indicators provide information that can easily be understood by policy makers and stakeholders.

Recent European jurisdiction on Environmental Impact Assessment procedures and physical planning has shown that it is compulsory to take into account the political and/or legal protection status of areas or species in the process of decision-making. If legislation concerning species protection is neglected, this can result in serious delay or even prohibition of implementation of planned measures. Since financial means and sufficient time to gather additional field data are often lacking, assessment of biodiversity should preferably also be possible on the basis of flora, fauna and landscape ecological data already available. Methodologies for assessment of biodiversity of the riverine areas of the Rhine and Meuse must be applicable in a transnational context, and account for actual as well as potential situations. Another demand is compatibility with hydraulic and hydromorphological models, and the possibility to use input data from various levels of scale.

The objective of this paper is to present the development and application of such a model, a transnational model for assessment of impacts of physical reconstruction on biodiversity. This model is conceived on the basis of protected and endangered species of north-western Europe, characteristic of the Rhine and Meuse. The study contributed to the further elaboration of an existing ***Spreadsheet Application For Evaluation of BIOdiversity*** (BIO-SAFE, Lenders ***et al.***, 2001) into a transnational river management tool suitable for the rivers Rhine and Meuse in the Netherlands, Germany, France and Belgium.

2 Materials and methods

2.1 Conceptual framework of BIO-SAFE

The conceptual framework of BIO-SAFE concerns the confrontation of physical reconstruction with policy and legislation regarding protection of biodiversity and ecological improvement. The basis of BIO-SAFE is therefore formed by the national and international conservation policy and legislation concerning endangered and protected species characteristic of river ecosystems (left hand part of Figure 1). Values are assigned to each species on the basis of its status according to national Red Lists and international directives and conventions. This assignment of values to species enables assessment of an actual situation of a floodplain on the basis of data on species presence in that particular area. By describing the species' habitat demands using a landscape ecological classification typology, values can also be assigned to patches in the floodplain, ***e.g.*** ecotopes. 'Ecotope' is a generally accepted term for the geographical part of an ecosystem (Neef, 1967; Haase, 1989). Klijn & Udo de Haes (1994) define an ecotope as a spatial unit of a certain extent, which is homogeneous regarding vegetation structure, succession stage and the main abiotic site factors that are relevant to plant growth. Ecotopes are potential habitats (Harper ***et al.***, 1995) for species. The linkage of species to specific ecotopes is the basis for valuation of the biodiversity potential in a particular area.

Flood risk reduction measures alter the physical and biological conditions of a floodplain and, as a result, the potential value of that floodplain to biodiversity (right hand part of Figure 1). Input data are flora and fauna data and/or ecotope data from field surveys carried out within the framework of reconstruction projects or from existing databases. Ecotope data of various levels of scale can be used (see the next section). When different alternatives for reconstruction are described in terms of ecotopes, these alternatives can be assessed. Comparison of the situation before reconstruction, or a reference scenario with the alternatives for reconstruction (potential situations) results in an assessment of the impacts of physical reconstruction on biodiversity that may be expected.

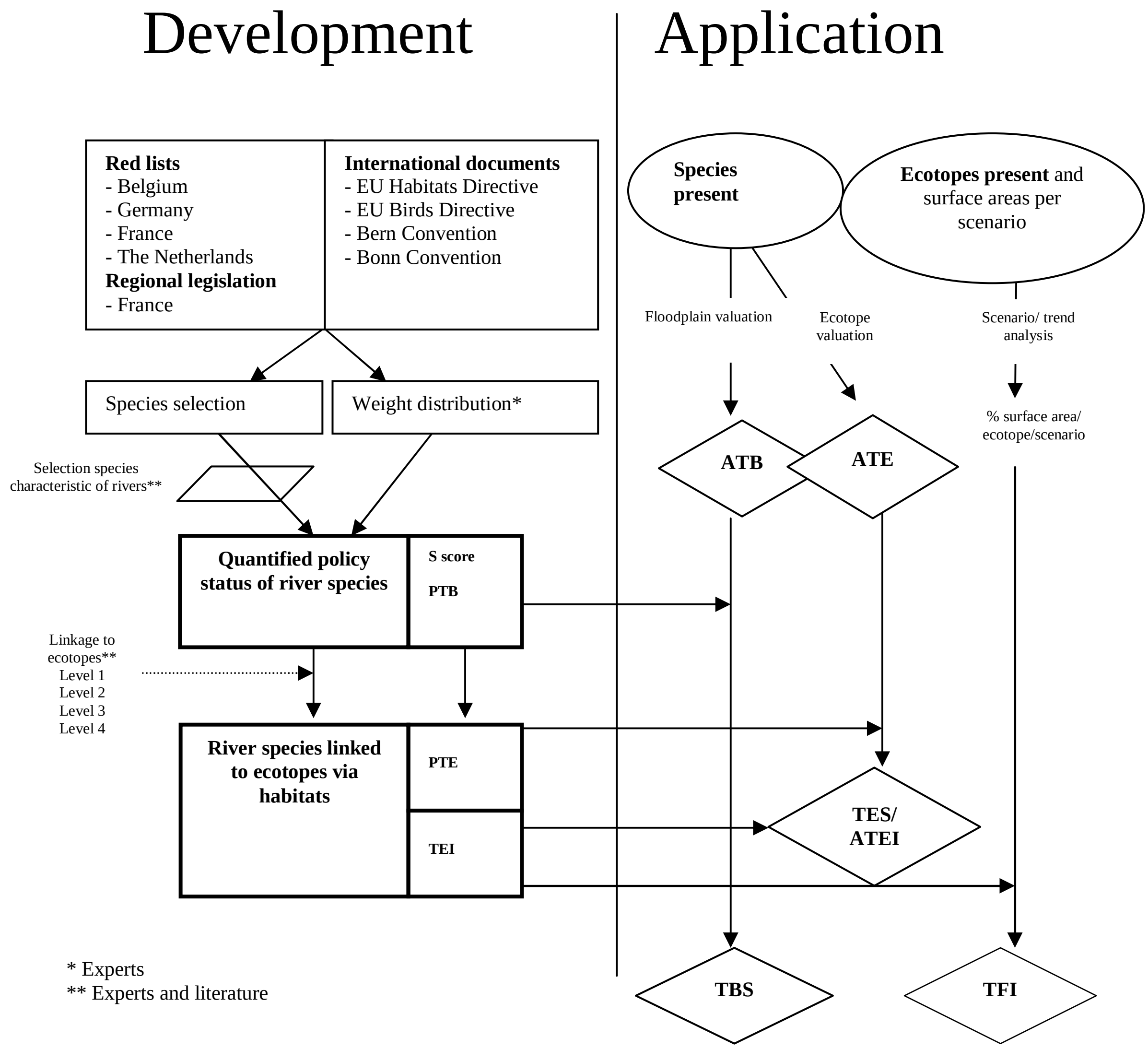

Figure 1. Selection and value assignment procedure followed for relevant species and ecotopes in developing BIO-SAFE (left hand side), and index calculation procedure for application (right hand side). Bold boxes: data integrated in the model. For explanation, see text.

2.2 Selection of species and linkage to ecotopes

The first step in constructing BIO-SAFE comprised the selection of species. Species to be selected had to be 1) relevant in terms of policy or legislation, and 2) indigenous to and characteristic of riverine areas. The development of BIO-SAFE was based on species characteristic of rivers and floodplains, but the principles of the method can easily be applied to other ecosystems. Relevancy for policy and legislation was made operational along two lines (Figure 1). The first line relates to species designated as ‘protected’ or ‘special attention’ species in international treaties and directives. This selection included bird species mentioned in Annex I of the EU Birds Directive (Council Directive 79/409/EEC), species mentioned in Annexes II, IV or V of the EU Habitats Directive (Council Directive 92/43/EEC), species mentioned in Appendices I or II of the Bonn Convention (Intergovernmental Treaty, Bonn

1.XI.1983) and species mentioned in Appendices I or II of the Bern Convention (Council of Europe, Bern 19.IX.1979, European Treaty Series/104).

The second line relates to nationally endangered species. In this study, this concerned species meeting national Red Data list criteria according to the World Conservation Union (IUCN, 1993, 1994, 2001). The taxonomic groups involved are higher plants (Spermatophyta), dragonflies and damselflies (Odonata), butterflies (Lepidoptera), fish (Pisces), amphibians and reptiles (Herpetofauna), birds (Aves) and mammals (Mammalia). Species belonging to other taxonomic groups either do not meet the political selection criteria or are not characteristic of rivers or floodplains (Lenders ***et al.***, 2001). No French Red Lists for plants were applicable to the Meuse valley. Plant species selected for France were therefore by way of exception taken from documents concerning the legal protection of species at the regional level. The sources of the Red Lists are given in De Nooij ***et al.*** (2001).

Determination of whether a species is characteristic of river ecosystems of Rhine and Meuse and which ecotopes are used as habitat was based on: 1) ecological literature describing species characteristics, habitats and historical-geographic distribution maps of species, and 2) expert judgement. A questionnaire was designed separately for the Netherlands, Germany, France and Belgium and sent to specialists in the field of the relevant species groups and/or riverine ecology of these four countries. Each expert only selected species concerning his own country. For the description of the habitats of the species selected by the experts, each questionnaire included an ecotope typology for the rivers Rhine and Meuse (Figure 1). Ecotopes at four levels of scale were used: 1:100,000 (units of the third level of the CORINE land cover classification (EC, 1992)); 1:50,000 (units of the biotope typology that is currently developed by the International Committee for the Protection of the Rhine (ICPR, derived from DIREN Alsace, 2000)); 1:25,000 and 1:10,000 (units of the Dutch River Ecotope System (Rademakers & Wolfert, 1994)). Furthermore, the ecotopes at 1:10,000 were linked to phyto-sociological units. This was done to enable BIO-SAFE to use input data in phyto-sociological terms. The ecotope typology applied allows aggregation of assessment results on low levels of spatial scales to higher spatial scales and the use of input data of various scales. De Nooij ***et al.*** (2001) presented a complete overview of the questionnaire, the expert panel and their institutes, the literature applied and the ecotope typology used.

The results of the species selection process are given in Table 1. The end selection consists of 257, 171, 160 and 173 species for the Netherlands, Germany, France and Belgium, respectively. The total number of different species is 486. In many cases there is an overlap between the countries regarding the species selected. Most species were linked to more than one ecotope. The database can, as an integral part of the BIO-SAFE model, be found on a CD-ROM distributed by DES/KUN (2001).

Table 1. Numbers of species meeting the selection criteria, per taxonomic group, per country.

Taxon	Species of Red Lists				Species of HD, BD, CBern, CBonn[4]				RP	End selection			
	NL	G[1]	F[2]	B	NL	G	F[2]	B	F	NL	G	F[2]	B
Higher plants	499[5]	1580	56	539	10	527	28	18	11	136	60	12	90
Birds	57	197	142[3]	107	220	222	155	220	-	60	58	113	38
Reptiles and Amphibians	15	28	7	16	23	25	18	23	-	9	11	7	4
Mammals	21	69	17	32	23	33	31	33	-	9	11	7	5
Fish	24	64	14	34	26	26	7	26	-	20	17	10	16
Butterflies	47	21	15	82	5	5	5	5	-	17	9	7	15
Dragon- and Damselflies	27	62	2	41	17	17	7	18	-	6	5	4	5
Total	690	2021	253	851	324	855	251	343	11	257	171	160	173

HD: Habitats Directive, BD: Birds Directive, CBern: Convention of Bern, CBonn: Convention of Bonn, RP: Regional protection.
[1] Including the Red List of the German federal states Baden-Württemberg, Hessen, Nordrhein-Westfalen and Rheinland-Pfalz.
[2] North-eastern France.
[3] Concept of a regional Red List for birds by the University of Metz.
[4] Only the species of the Rhine and Meuse basins.
[5] Proposal for the official Red List for higher plants by Van der Meijden ***et al.*** (2000).
- Not considered.

2.3 Assignment of values to species and ecotopes

The next step in constructing BIO-SAFE was the assignment of values to the species selected: quantifying the relative differences in relevance to policy between the species. Relative weights were assigned to the same criteria used to select the species (Figure 1) by an international expert panel. The expert panel consisted of Dutch, French, German and Belgian professionals in the field of species protection. The panel was sent a questionnaire in which they were asked to distribute a fixed number of 40 points between the valuation criteria. They were told that the weight assigned should reflect the relative importance of each instrument in nature policy in their country. As a result, mean weights were calculated separately for the Netherlands, Belgium, Germany and France (see Table 2). The questionnaire and the expert panel are given in De Nooij ***et al.*** (2001).

Applying the valuation criteria, leads to the assignment of a ***Species-specific*** score (S-score) to each species selected by summation of the values assigned to the criteria applicable to a species (Lenders ***et al.***, 2001). To the White stork ***(Ciconia ciconia)*** for example, applying the criteria led to an S-score of 28.4 being assigned in the Netherlands, 22.7 in Germany, 30.6 in France and 28.0 in Belgium (Table 2). In order to make it possible to calculate taxonomic group level biodiversity assessments, the S-scores of species belonging to a particular taxonomic group were summed to yield a Potential Taxonomic group Biodiversity (PTB) constant (Figure 1). This constant reflects the maximum score possible for the taxonomic group involved.

On the basis of the species habitat requirements, values are assigned to ecotopes as well. For each ecotope type, the S-scores assigned to the species linked to the ecotope are summed per taxonomic group, yielding a ***Potential Taxonomic group Ecotope*** (PTE) constant (Figure 1), i.e. the maximum score for an ecotope from the viewpoint of a particular taxonomic group. Subsequently, this PTE constant is related to the PTB constant, resulting in a ***Taxonomic group Ecotope Importance*** constant (TEI), ranging from 0 to 100 per ecotope type

(Figure 1, equation 1). The TEI reflects the importance of an ecotope with respect to conservation values. ***Taxonomic group Ecotope Importance*** constants can be calculated for each country specifically, on four levels of spatial scale, on the basis of different combinations of the criteria. Table 3 presents an example where all valuation criteria were applied.

$$TEI = PTE * 100 / PTB \qquad [1]$$

TEI = ***Taxonomic group Ecotope Importance*** constant
PTE = ***Potential Taxonomic group Ecotope*** constant
PTB = ***Potential Taxonomic group Biodiversity*** constant

The relative importance value of each ecotope in Table 3 often differs greatly between the taxonomic groups within each country. There are also large differences between the countries. For example, it appears that ecotopes that are almost never flooded, the so-called high-water-free ecotopes, are much more valuable in the Netherlands than in the other countries.

Table 2. Valuation criteria applied and their mean weights and standard deviation according to an international expert panel.

Criteria	Weight distribution				Comments
	NL n=17	G n=6	F n=7	B n=5	
Red Lists*	6.9 ± 3.2	10.8 ± 4.1	5.3 ± 1.5	5.4 ± 2.7	(IUCN-criteria ‘extinct’, ‘critical’, ‘endangered’ or ‘vulnerable’ ‘susceptible’)
Regional protection	-	-	9.7 ± 1.3	-	Sub-national level, applied to plants only
Red lists of states*	-	8.8 ± 1.2	-	-	
Bern Convention (1)*	5.7 ± 2.4	2.0 ± 1.4	3.6 ± 1.4	4.6 ± 2.2	
Bonn Convention (2)*	5.6 ± 2.7	2.2 ± 1.3	3.7 ± 1.6	5.2 ± 1.8	
EU Birds Directive (3)*	10.1 ± 2.8	7.7 ± 2.2	8.3 ± 1.8	12.8 ± 3.7	EU-Birds Directive and EU-Habitats Directive are complementary. EU-Birds Directive applicable to birds only; EU-Habitats Directive applicable to all other species.
EU Habitats Directive (4)	11.6 ± 3.1	8.5 ± 1.9	9.4 ± 0.8	12.0 ± 2.1	
- Annex II only	11.6	8.5	9.4	12.0	
- Annex IV only	5.8	4.3	4.7	6.0	
- Annex V only	2.9	2.1	2.4	3.0	
- Annexes II <u>and</u> IV	11.6	8.5	9.4	12.0	
- Annexes II <u>and</u> V	11.6	8.5	9.4	12.0	
- Annexes IV <u>and</u> V	5.8	4.3	4.7	6.0	
- Annexes II, IV <u>and</u> V	11.6	8.5	9.4	12.0	

n: number of respondents.
1: Appendices I and II: strictly protected flora and fauna species respectively.
2: Appendix I: migratory species whose immediate protection is required; Appendix II: migratory species whose conservation and management should be covered by means of transnational agreements.
3: Annex I: species that are subject of special conservation measures concerning their habitat in order to ensure their survival and reproduction in their area of distribution.
4: Annex II: species whose conservation requires the designation of special areas of conservation; Annex IV: species in need of strict protection; Annex V: species whose taking in the wild and exploitation may be subject to management measures.
*: Criteria that apply to the White stork (***Ciconia ciconia***).

Table 3. ***Taxonomic group Ecotope Importance*** constants (TEI; 0-100) for the Netherlands, Germany, France and Belgium reflecting the importance of ecotopes at level 3 (1:25,000) with respect to conservation values for species belonging to a particular taxonomic group.

Ecotope	Taxonomic group Ecotope Importance															
	NL				G				F				B			
	HP	BI	FI	DD	HP	BI	FI	DD	HP	BI	FI	DD	HP	BI	FI	DD
Deep summer bed	0	13	73	0	0	12	87	0	0	21	81	0	0	7	85	0
Shallow summer bed	1	10	85	61	0	8	68	37	0	20	89	0	2	3	85	75
Side channel	1	25	45	38	5	8	68	37	8	25	34	0	2	11	9	63
Beach, Bank, Bar	14	13	47	69	40	37	63	37	17	35	11	0	9	16	9	37
Shallow waters	3	64	44	31	15	32	4	0	0	34	49	0	19	25	15	75
Lake	1	40	27	0	12	21	0	0	8	39	27	0	3	30	0	50
Herbaceous marsh	19	42	3	23	39	34	0	0	0	40	0	0	6	32	0	0
Marsh grassland	5	32	0	61	32	29	0	0	0	49	0	0	1	35	0	12
Moist grassland	11	33	0	61	34	25	0	0	17	46	0	0	8	21	0	12
Levee pastures	43	8	0	0	0	10	0	0	8	27	0	0	20	7	0	0
High-water-free grassland	28	7	0	61	0	12	0	0	0	24	0	0	13	7	0	12
High-water-free herbaceous area	6	7	0	61	0	6	0	0	8	18	0	0	8	3	0	12
Shrubs in floodplain	1	9	0	69	18	0	0	0	0	11	0	0	3	2	0	88
Shrubs on levee	4	1	0	69	0	14	0	0	0	7	0	0	5	0	0	12
Softwood alluvial forest	1	11	0	8	9	4	0	0	0	5	0	0	5	1	0	75
Hardwood alluvial forest	2	5	0	8	7	0	0	0	0	6	0	0	6	1	0	75
Forested levee	5	8	0	8	0	18	0	0	0	6	0	0	6	1	0	0
High-water-free forested area	3	5	0	69	0	5	0	0	0	6	0	0	5	1	0	25
River dune	17	0	0	0	7	3	0	0	17	12	0	0	14	3	0	0
Gravel deposit	1	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0

HP: Higher Plants, BI: Birds, FI: Fish, DD: Dragon- and Damselflies.

2.4 Index and score calculation

Data on actual presence of species in a particular area can by means of BIO-SAFE be used to calculate two types of indices. The first type concerns biodiversity indices at the taxonomic group level. For this purpose the S-scores of the species ***actually present in an area*** are summed, yielding an ***Actual Taxonomic group Biodiversity*** score (ATB score; Figure 1). This score reflects the actual value of the area per taxonomic group. The Potential Taxonomic group Biodiversity (PTB) constants and the Actual Taxonomic group Biodiversity (ATB) scores can be used to calculate ***Taxonomic group Biodiversity Saturation*** (TBS) indices ranging from 0 to 100 (Figure 1: equation 2). The TBS indices offer insight into the degree to which the maximum expected biodiversity value per taxonomic group has actually been achieved in a particular area. An aggregated biodiversity index is calculated by averaging the TBS indices for taxonomic groups for which (sufficient) field data are present.

The second type of index reflects the actual value of each ecotope present in an area. For each taxonomic group, the S-scores assigned to the ecotopes linked to species ***actually present*** were summed up per ecotope type, yielding an ***Actual Taxonomic group Ecotope*** score (ATE score). This ATE score was related to the PTE constant, resulting in a ***Taxonomic***

group Ecotope Saturation (TES) index per ecotope type, ranging from 0 to 100 (Figure 1: equation 3). Multiplication of this index with the ***Taxonomic group Ecotope Importance*** (TEI) constant of that particular ecotope type yields a score that offers insight into the significance of that ecotope type for a specific taxonomic group in the area studied (Figure 1: equation 4): the ***Actual Taxonomic group Ecotope Importance*** score (ATEI score).

In order to be able to compare the effects of different reconstruction alternatives (scenarios) on biodiversity potentials, for each taxonomic group the ***Taxonomic group Ecotope Importance*** constants per ecotope type were multiplied by the relative surface area, derived from GIS maps, of that particular ecotope type to be realised within each scenario. This is also done for a reference scenario with no measures taken. For each scenario these products were summed per taxonomic group, thus offering insight into the significance of each scenario for that particular taxonomic group. This results in the ***Taxonomic group Floodplain Importance*** score (TFI, equation 5). The ***Taxonomic group Floodplain Importance*** score (TFI) can be summed up per scenario yielding the ***Floodplain Importance*** score (FI, equation 6). A further explanation of the calculation and use of the indices and scores is given in De Nooij ***et al.*** (2001).

BIO-SAFE was implemented in a spreadsheet application (MS Excel 97®). Separate tools were constructed for the Netherlands, Germany, France and Belgium, thus allowing biodiversity assessment on the basis of political and legal criteria using the species sets and valuation criteria applicable to each country. After an easy-to-use standardised spreadsheet form has been completed, the program calculates all relevant indices and scores.

$$\boldsymbol{TBS = ATB * 100 / PTB} \qquad [2]$$

TBS = ***Taxonomic group Biodiversity Saturation*** index
ATB = ***Actual Taxonomic group Biodiversity*** score
PTB = ***Potential Taxonomic group Biodiversity*** score

$$\boldsymbol{TES = ATE * 100 / PTE} \qquad [3]$$

TES = ***Taxonomic group Ecotope Saturation*** index
ATE = ***Actual Taxonomic group Ecotope*** score
PTE = ***Actual Taxonomic group Ecotope*** constant

$$ATEI = TES * TEI / 100 = ATE * 100 / PTB \qquad [4]$$

ATEI = ***Actual Taxonomic group Ecotope Importance*** score
TES = ***Taxonomic group Ecotope Saturation*** index
TEI = ***Taxonomic group Ecotope Importance*** constant
ATE = ***Actual Taxonomic group Ecotope*** score
PTB = ***Potential Taxonomic group Biodiversity*** constant for the investigated taxonomic group

$$TFI_y = \Sigma \left((SA_{ecotope(x)} / SA_{floodplain}) * TEI_x \right) \qquad [5]$$

TFI_y = ***Taxonomic group Floodplain Importance*** score for taxonomic group y
n = total number of ecotopes present in the floodplain
$S_{ecotope(x)}$ = surface area of ecotope x present in the floodplain
$S_{floodplain}$ = surface area of the floodplain
TEI_x = ***Taxonomic group Ecotope Importance*** constant for ecotope type x

$$FI = \Sigma\ TFI_y \qquad [6]$$

FI = ***Floodplain Importance*** score
n = total number of relevant taxonomic groups y
TFI_y = ***Taxonomic group Floodplain Importance*** score for taxonomic group y

2.5 Application of BIO-SAFE

BIO-SAFE can be applied for the purpose of (a) valuation of the actual situation (at the level of taxonomic groups, ecotopes and at the floodplain level), (b) evaluative analysis of different scenarios or designs for reconstruction of a floodplain, allowing assessment of impacts of different reconstruction measures and a ranking of reconstruction alternatives according to their value for biodiversity conservation (on the level of taxonomic groups and on the floodplain level), (c) valuations of ecotopes and transitions between ecotopes and (d) trend analysis, showing biodiversity value patterns in time.

The area of application comprised the riverine areas of the river basins of the Rhine and the Meuse (Figure 2). Riverine areas were defined as the areas between the winter dikes, or the area flooded during the maximum high water level, of the main branches of Rhine and Meuse. Mountainous and estuarine zones were not included in the study. Along the longitudinal axis of the river Rhine this comprised the Rhine from the Swiss-German border to Schoonhoven (Nederrijn-Lek), Gorinchem (Waal) and Kampen (IJssel) in The Netherlands. For the river Meuse the area of application comprises the Meuse from the source of the river to the intertidal area of the Biesbosch in The Netherlands**.**

BIO-SAFE was applied to a number of floodplain areas in the Netherlands, Germany, Belgium that have been or will be subjected to flood prevention measures that influence the physical (abiotic) characteristics of the area (for locations see Figure 2). In this study all the valuation criteria were used for application of BIO-SAFE. Valuation of the actual situation was done for all cases, scenario analysis concerned Rijnwaarden, the Common Meuse and Vynen/Rees. Trends were analysed for Rijnwaarden and Gameren. These floodplains are briefly described below. More detailed information about the floodplain areas, the data sources and the reconstruction scenarios can be found in De Nooij ***et al.*** (2001).

Plans for reducing flooding risks in combination with ecological rehabilitation for the Rijnwaarden floodplain area (1,100 ha) have been prepared by the Dutch Institute for Inland Water management and Waste Water Treatment, RIZA (Van Rooij & Kappers, 1998, VISTA/Staring Centre, 1998). Species presence data were obtained from the Gelderland Provincial Authorities, NGO’s, field reports and distribution atlases. A predicted ecotope trend was taken from Van der Lee ***et al.*** (2001).

The Gameren floodplain (144 ha) is a nature reserve, where three secondary channels have been created for the purpose of a combined flood risk reduction and nature rehabilitation plan. Data on species and ecotope presence were obtained from NGO’s and the Dutch Institute for Inland Water management and Waste Water Treatment.

The Vynen/Rees area (120 ha) is partly a nature reserve and bears high ecological potentials. Present vegetation and fauna data and predicted ecotope were obtained from the German Federal Institute of Hydrology (BfG).

Plans for the Common Meuse (2365 ha), aiming at reducing flooding risks in combination with ecological rehabilitation, were prepared by the Institute of Nature Conservation (Van Steertegem, 2000). Data on species presence were obtained from the Flemish Institute of Nature Conservation and from additional reports.

Data on species and ecotope presence in the Mouzay floodplain (570 ha) were made available by the University of Metz and the French Society for Odonatology (Societé Française d’Odonatologie, SFO).

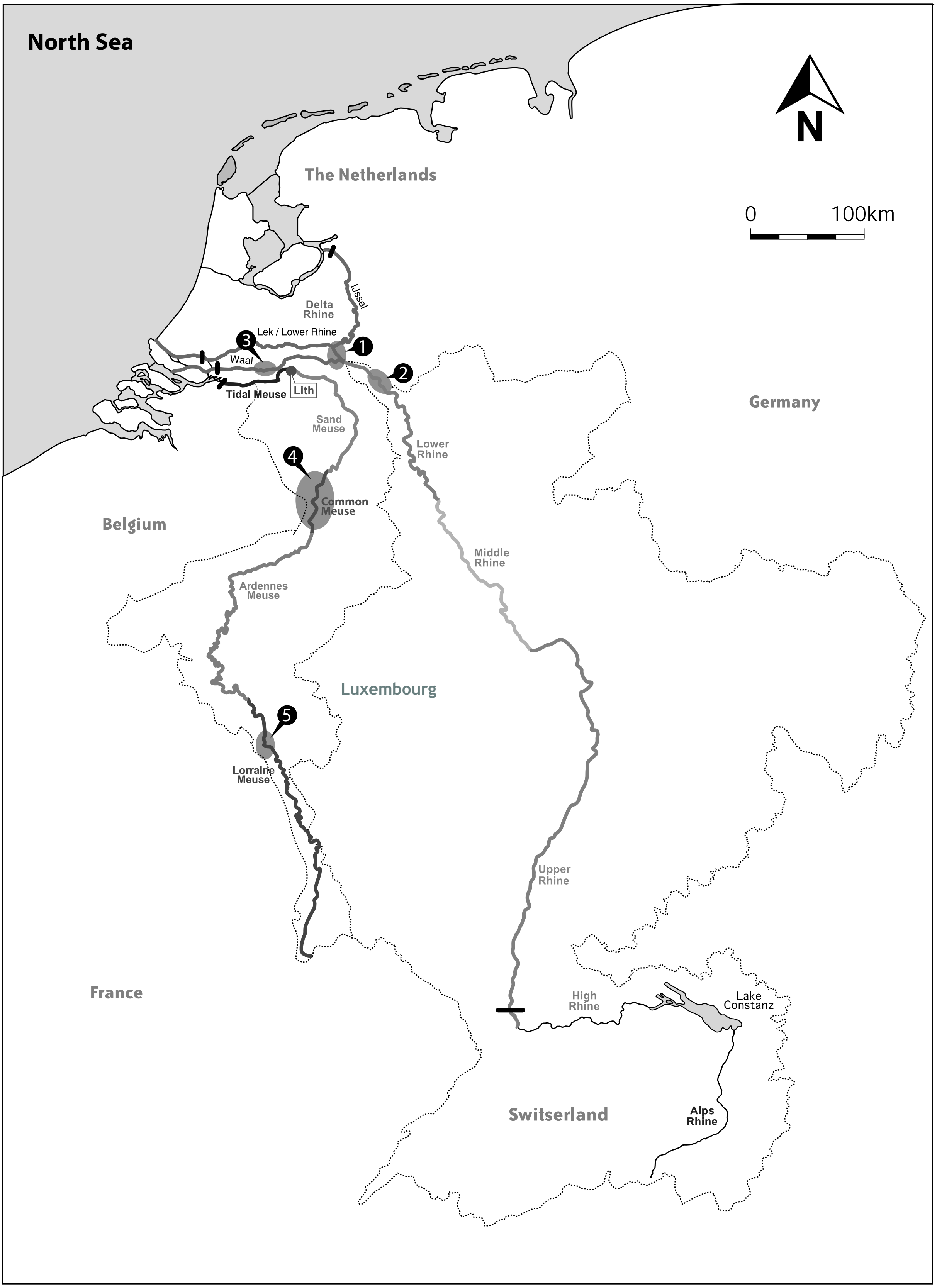

Figure 2. The area of application and the locations of the case study areas. 1: Rijnwaarden, 2: Vynen-Rees, 3. Gameren, 4: Common Meuse, 5: Mouzay. The bars indicate the longitudinal delineation of the study area.

3 Results

3.1 Actual situations

Table 4 shows that the saturation indices calculated differ greatly among the taxonomic groups and between the areas. In all cases studied, the saturation indices of birds are the highest. Saturation values of higher plants are high in both sites along the Meuse, and much lower in the sites along the Rhine. Part of the differences between taxonomic groups may be due to incomplete distribution surveys of some groups (mammals, butterflies, damselflies and dragonflies, and fishes). It appears that the present value of the Rijnwaarden floodplain can be attributed largely to birds, closely followed by mammals and reptiles and amphibians. Remarkably low are the indices for higher plants and, especially, dragonflies and damselflies. In the floodplain Vynen/Rees, birds are by far the most important group. Results for the common Meuse show that the area is important for higher plants and birds. Available data for the Mouzay area indicated that mammals and dragon- damselflies are highly underrepresented.

Table 4. ***Taxonomic group Biodiversity Saturation*** indices (TBS; 0-100) for seven taxonomic groups in all case study areas.

Taxonomic group	Taxonomic group Biodiversity Saturation			
	Rijnwaarden	Vynen/Rees	Common Meuse	Mouzay
Higher plants	19.2	6.0	58.2	50.0
Birds	62.9	48.9	58.4	56.6
Reptiles and amphibians	42.0	0.0	-	36.3
Mammals	52.2	-	-	0.0
Fish	24.1	-	22.6	-
Butterflies	0.0	-	-	-
Dragonflies and damselflies	8.5	-	-	0.0
Biodiversity Saturation (mean value)	29.8	18.3	46.4	28.6

- not assessable due to lack of data

3.2 Scenario analysis

The scenarios for the Rijnwaarden floodplain, the Vynen/Rees floodplain and the Common Meuse area are compared in Table 5. In each case the potential of the reference scenario is given in absolute numbers (***Taxonomic group Floodplain Importance*** score, (TFI)) and ***Floodplain Importance*** score (FI)), and the potentials of the reconstruction scenarios are expressed as the difference compared to the reference in percentages.

The results concerning Rijnwaarden show that the potentials for higher plants and insects selected for BIO-SAFE strongly increase in all reconstruction scenarios (Table 5). The scenarios aimed at low influence of river dynamics (R1 and R2) in the floodplain offer the best opportunities for all groups except plants. As far as birds are concerned, it may be noticed that especially the high dynamics scenarios (R3 and R4) result in considerably lower potentials than the low dynamics scenarios.

Scenario analysis for the Common Meuse shows that the two scenarios that aim at development of new ecotopes (C2 and C3) are positive for most groups, especially for herpetofauna. However, in absolute terms, the effects on plants and butterflies are so negative that only the most natural scenario (C3) has positive overall effects.

The scenario analysis for Vynen/Rees shows that the scenario calculated for an increase of the water levels (V1) increases potentials for all species groups except mammals; the scenario for a decrease in water levels (V2) has negative impacts on all species groups except birds.

Table 5. Potentials of the (reconstruction) scenarios developed for the Rijnwaarden area, the Vynen/Rees floodplain and the Common Meuse floodplain. %: relative change compared to reference scenario.

Case study	Taxonomic group Floodplain Importance score							
	Higher plants	Birds	Reptiles and amphibians	Mammals	Fish	Butterflies	Dragon- and damselflies	Total (FI)
Rijnwaarden								
R0	355	2227	4993	3998	1256	346	767	13941
R1	+ 79%	+ 36%	+ 33%	+ 21%	+ 11%	+ 278%	+ 208%	+ 45%
R2	+ 84%	+ 46%	+ 41%	+ 28%	+ 29%	+ 187%	+ 173%	+ 49%
R3	+ 186%	+ 8%	+ 28%	+ 1%	+ 4%	+ 95%	+ 63%	+ 22%
R4	+ 176%	+ 7%	+ 23%	+ 1%	+ 18%	+ 88%	+ 75%	+ 22%
Common Meuse								
C0	755	626	25	2595	254	1952	2264	8471
C1	- 7%	+ 1%	- 17%	+ 1%	+ 1%	- 8%	+ 1%	- 2%
C2	- 56%	+ 21%	+ 1776%	+ 2%	+ 25%	- 54%	+ 9%	- 7%
C3	- 70%	+ 30%	+ 3526%	+ 17%	- 36%	- 66%	+ 31%	+ 4%
Vynen/Rees								
V0	206	662	1866	1939	478	960	121	6231
V1	+ 14%	+ 5%	+ 8%	0%	+ 16%	+ 7%	+ 19%	+ 6%
V2	- 8%	0%	- 32%	- 1%	- 12%	- 4%	- 11%	- 12%

FI: ***Floodplain Importance*** score.
R0: Reference scenario (no measures taken); R1: Low Dynamics Scenario for a 16,000 $m^3 s^{-1}$ design discharge at Lobith; R2: Low Dynamics Scenario for a 18,000 $m^3 s^{-1}$ design discharge at Lobith; R3: High Dynamics Scenario for a 16,000 $m^3 s^{-1}$ design discharge at Lobith; R4: High Dynamics Scenario for a 18,000 $m^3 s^{-1}$ design discharge at Lobith.
C0: Reference scenario (no measures taken); C1: Agricultural land maintained as in a present situation and conservation of present natural values; C2: Development of nature and a decrease of agriculture; C3: Maximisation of natural processes.
V0: Reference scenario (no changes in water level); V1: 50 cm increase of the mean water level; V2: 50 cm decrease of the mean water level.

3.3 Trend analysis

Analysis of a predicted ecotope trend for Rijnwaarden (Van der Lee ***et al***, 2001) using the ***Taxonomic group Floodplain Importance*** score (Figure 3) shows that the overall pattern is a steady increase of biodiversity, until 15 years after reconstruction. For plants and butterflies, it can be seen that the potentials drop steeply after a small initial rise. The potentials for dragon- and damselflies first drop and then rise slightly.

The trend data of the period 1998-2000 of higher plants and fish in the Gameren floodplain show a decrease of saturation of higher plants from 6.4 to 4.3. The saturation of fish increased from 9.1 to 14.4, in the period 1999-2000.

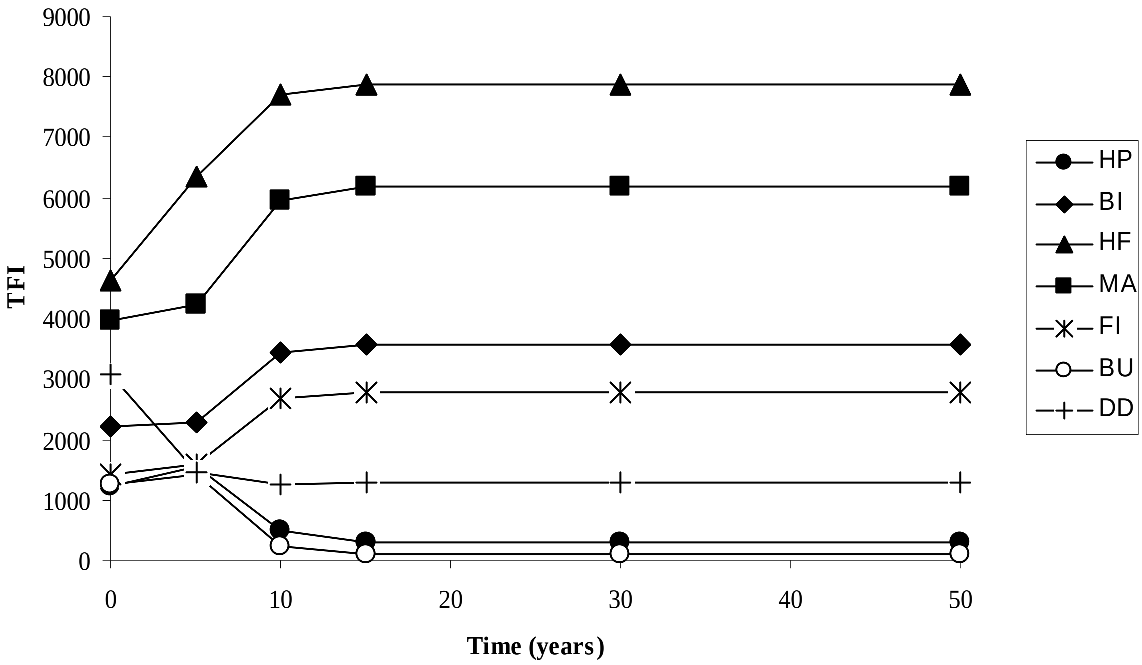

Figure 3. Analysis of a predicted trend for the Rijnwaarden floodplain (TFI: ***Taxonomic group Floodplain Importance*** score, potentials per taxon each scenario, HP: Higher Plants, BI: Birds, HF: Reptiles and Amphibians, MA: Mammals, FI: Fish, BU: Butterflies, DD: Dragon- and Damselflies).

4 Discussion

4.1 Added value of the model

Attuning biodiversity conservation and flood risk reduction measures is a major issue in applied ecology and planning. In several planning procedures it is compulsory to take (habitats of) protected species into consideration. BIO-SAFE offers the opportunity to present politically and legally relevant indices on the level of taxonomic groups and an aggregated level for ecotopes or floodplains and reconstruction designs. The method links ecological knowledge with (inter)national nature conservation policy goals, by providing insight into ecologically relevant parameters from the viewpoint of protected and endangered species. The frequently used biodiversity indices (for example Shannon, Simpson's, Berger-Parker Dominance) are often poorly understood by policy makers and cannot give insight into the potentials of taxonomic groups and ecotopes (Feest, 2000), or the consequences of reconstruction measures. Therefore, BIO-SAFE can be regarded as complementary to traditional methods. Assessments are possible on the basis of available data on species and/or ecotopes and can be carried out at four levels of spatial scale. An assessment using BIO-SAFE can be based on EU directives, international treaties on species protection, national Red Lists concerning endangered species and on all these political and legal criteria simultaneously.

BIO-SAFE was constructed as a transnational and river specific instrument that uses the policy status and habitat demands of between 160 and 254 river characteristic species, that are for a large part dependent on riverine habitats in floodplains. Because the selection of riverine species is partly based on expert judgement, it carries a subjective touch. However, there seems to be no way to renounce this subjectivity because there are no data available that allow a selection based on mathematical or logical algorithms. Regarding any selection of species, it is an important notion that, since vascular plants and vertebrates together make up less than 10% of known biodiversity, any particular set of species is an extremely small fraction of the biota of any one place (Franklin, 1993). Conservancy legislation seems to aim at large and appealing species only. There are, for example, no representatives of taxonomic groups like lower plants and only a few macroinvertebrates protected by national and international legislation. Some of these taxonomic groups may be far more relevant ecological indicators (***e.g.***, Van den Brink ***et al.***, 1996; Lenders ***et al.***, 2001).

Although subjective, valuation using conservation status is often applied in policy, research and model development (Freitag ***et al.***, 1997; Oertli ***et al.***, 2002; Ten Brink ***et al.***, 2001). In this study, a panel of specialists composed of scientists, policy makers and conservationists carried out the value-assignment. The number of panellists from Germany, France and Belgium was low, and the value-assignment in these countries might not be representative. The outcomes of the questionnaire show more or less the same pattern of weight distribution over the various instruments in all the four countries. However, comparisons between the different approaches towards species conservation of the different countries suggest that regional legislation is most important in France, Red Lists play a dominant role in Germany, whereas European legislation is given highest weights in Belgium and the Netherlands. In consideration of the European Water Framework Directive, international harmonisation of the different approaches is recommended (***e.g.*** development of ecological targets at the river basin scale).

The use of ecotopes provides a basis for assessing biodiversity at the landscape level in addition to the species level, and gives a clear insight into the spatial consequences of species protection. The hierarchical ecotope typology was constructed by integration of landscape ecological typologies that are well established and relevant in the context of river management in northwestern Europe. Although only one of the possible approaches towards biodiversity, a hierarchical approach provides a theoretical basis for understanding biodiversity patterns in river systems as spatially nested in the catchment structure and being a result of the processes that play a role on and across various spatial and temporal scales. Klijn & Udo de Haes (1994) investigated the possibilities of a hierarchical approach to ecosystems and its implications for ecological land classification and exemplify that the approach is particularly valuable as a comprehensive tool for scientific analysis on behalf of environmental policy. The ecotope typology is the component of the model that allows up- and down scaling and makes it compatible with hydrological and morphological models and common descriptions of river systems. The use of an ecotope typology also had some disadvantages. BIO-SAFE assigns a species to a set of ecotopes, whereas animals typically exhibit complex habitat requirements in space and time, and not every ecotope is equally important to a species. The information required to account for this complexity in a model is however not available for most species. The predictive power of BIO-SAFE is therefore limited to the use of species-ecotope relationships for an estimate of the effects of changes of ecotope presence and surface area. The potential of ecotopes and floodplains is determined by a very complex set of factors that can vary from place to place and time to time (Leuven & Poudevigne, 2002). Prediction of ecotopes is not possible using BIO-SAFE. For this information the model must rely on other models (hydraulic models, succession models or ‘ecotope generators’ (Spence & Hickley, 2000; Guisan & Zimmerman, 2000)).

4.2 Use of the model

BIO-SAFE is recommended as a tool for various policy and management purposes such as determining the effectiveness of nature management measures, scenario studies for assessment of physical planning projects beforehand and monitoring and evaluation of such projects afterwards. BIO-SAFE can also be used for underpinning spatial planning reports and environmental impact assessments for large-scale activities in river basins, especially as regards the implementation of the EU Habitats Directive and the EU Birds Directive. BIO-SAFE should be used together with traditional biodiversity indices, population network analysis and detailed single species models for impact assessment (Foppen & Reijnen, 1998; Lenders ***et al.***, 1998a; Leuven & Poudevigne, 2002). BIO-SAFE deals with spatial issues, and it offers help in decision processes, therefore BIO-SAFE is well suitable for being integrated in a GIS-based decision support system for flood prevention measures. The application of BIO-SAFE demonstrated that the model could be used for different purposes, ***e.g.*** valuation of actual situations, scenario studies, trend analysis and ecotope valuation.

Valuation of actual situations using BIO-SAFE shows very quickly for which protected and endangered species groups an area already is important, which groups are underrepresented and hence which ecotopes should be conserved or developed. Application of BIO-SAFE to various flood risk reduction scenarios in the Netherlands, Belgium and Germany demonstrated that there are large differences between different reconstruction scenarios regarding impacts on biodiversity. Most reconstruction designs assessed showed an increase of biodiversity, but also strong negative effects were calculated. Using BIO-SAFE it is possible to rank scenarios according to their impacts on protected and endangered biodiversity. Furthermore, BIO-SAFE shows which ecotopes are important for which species groups, which can help optimisation of the design in the later stages. Valuation of ecotopes shows that there are large differences between the relative importance values for the species groups. Aggregation of this information leads to loss of information, and valuation of landscapes using habitat demands should therefore not be based on just one or two species groups. Application of BIO-SAFE to trend data showed that trend analysis can effectively detect the temporal patterns of biodiversity. This can be used retrospectively to assess the success of management or restoration measures, or to determine a suitable reference situation. Assessing predicted trends can help planning future management or reconstruction measures and can determine what the consequences of hydraulic and morphological developments are for characteristic species with policy and legislation relevance.

4.3 Future research

Valuation of nature can be done on the basis of the economic value of ecosystems (De Groot, 1992; Costanza ***et al.***, 1998) and from the ecological, ethical and aesthetic perspective (Swart ***et al.*** 2001). Further development of BIO-SAFE will concern ecological, legal and aesthetic aspects. In order to optimise the landscape ecological meaning of BIO-SAFE, it is possible to link plant species to a more detailed ecotope classification, ***e.g.*** Runhaar ***et al.*** (1987) in which 87 ecotope types are distinguished. This would enable BIO-SAFE to integrate ecologically relevant factors such as moisture regime, nutrient availability and acidity of the substratum. BIO-SAFE must be tested by means of application in a larger number of case studies. Values calculated on the basis of ecotope data only must be compared with observed data on flora and fauna presence. The model can be optimised by setting minimum required ecotope surface area thresholds, required for underpinning the estimation of potentials (Eiswerth & Haney, 2001). It can also be interesting to incorporate relationships between ecotope surface areas and biodiversity values for species groups (Oertli ***et al.***, 2002), which can be derived from case studies and existing literature.

Concerning the ethical perspective, it is necessary to study the legal consequences of species and habitat protection in order to make the model relevant in a strictly legal context. Analysis of jurisdiction must point out what the consequences of protective norms are for human activities within the landscape and how the different legal instruments relate to one another. As concerns valuation of biodiversity from the aesthetic perspective, it might be possible to incorporate lay people's valuation of landscape and species (Appleton, 1975; Van den Berg, 1999) in the valuation methodology. In the future we intend to tackle these issues in order to provide an instrument that makes it possible to assess biodiversity from various perspectives in order to base decisions on a multi-perspective approach.

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Chapter III

Complementarity and indicator function of BIO-SAFE

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Abstract

This study evaluates the effects of ecological rehabilitation on biodiversity in floodplains along lowland rivers in the Netherlands, using two different approaches. Species richness was compared with a policy and legislation based index, calculated by means of BIO-SAFE. BIO-SAFE is a valuation model that incorporates only protected and endangered river species, derived from Red Lists, the European Habitats directive, the European Birds directive and the conventions of Bern and Bonn. The study included higher plants, birds, mammals, herpetofauna, odonates and butterflies.

In some cases, biodiversity in the floodplains significantly increased after rehabilitation. However, the outcome of the evaluation differs per taxonomic group and approach. The results show positive temporal trends with respect to species richness of higher plants, herpetofauna and odonates, and to a lesser degree mammals and butterflies. The policy and legislation based index increased to some extent, but not significantly, for higher plants, herpetofauna and butterflies. Correlation between species richness and policy based values was found in all sites for birds, herpetofauna, odonates and, to a lesser degree, higher plants. For higher plants, positive as well as negative correlations were found, depending on the site.

Based on these results, we argue that an ecological approach and a policy based approach yield complementary information. Therefore, it is recommended to apply a multi-approach methodology in monitoring and evaluation of restoration projects.

1 Introduction

Floodplain rehabilitation aims at increasing biodiversity by creating a larger spatio-temporal heterogeneity of the riverine landscape (Ward ***et al.***, 1999). Since the late 1980's measures have been taken in the Netherlands that reintroduced various natural processes in floodplains. Examples of these processes are low-density grazing by large semi-wild herbivores, hydrodynamics, erosion and sedimentation, and succession (Cals ***et al.***, 1988; Nienhuis ***et al.***, 2002). Evaluation of rehabilitation measures requires assessment of the degree to which conversion of abiotic and biotic conditions of the river floodplain ecosystem into a more natural state has taken place. This study focuses on the biotic aspect, ***i.e.*** biodiversity. A distinction can be made between two approaches, 1) biodiversity in terms of ecological diversity models (e.g. the Shannon index, Menhinick's index or species richness; see Magurran, 1988) and 2) biodiversity in terms of characteristic species that are endangered and/or protected by law, and are therefore given special attention in policy-making and management (policy target species). Scientific biodiversity indices are difficult to interpret for policy makers whose primary interest lies with the achievement of policy goals.

The Convention on Biological Diversity (UNEP, 1992) gives a general framework for conservation and assessment of biodiversity, but offers no concrete indicators for policy based evaluation. On the other hand, in national and international documents such as Red Lists (IUCN, 2001), the conventions of Bern (Council of Europe, 1991) and Bonn (UNEP, 1979), and the Habitats directive (Council Directive 92/43/EEC) and Birds directive (Council Directive 79/409/EEC), endangered and/or protected flora and fauna species are identified. European jurisdiction shows that it is compulsory to take the legal status of these species into account in the process of decision-making. Most ecological diversity models require detailed and complete data on abundance and distribution of species (Magurran 1988, Ravera, 2001). In many cases these are not available and therefore biodiversity assessments are usually focussed on the number of species (richness) or on target species (Meffe & Carroll, 1997). Since outcomes of evaluations can have considerable implications, e.g. for fund allocation, the choice for a particular approach deserves keen attention.

This study evaluates the effects of ecological rehabilitation on biodiversity in floodplains along lowland rivers in the Netherlands, using two different approaches. Species richness (approach 1) was compared with a policy and legislation based index, calculated by means of BIO-SAFE (approach 2). BIO-SAFE is a valuation model that incorporates only protected and endangered river species, derived from Red Lists, the European Habitats directive, the European Birds directive and the conventions of Bern and Bonn (De Nooij ***et al.***, 2004; Lenders ***et al.***, 2001). The goal of this comparative study is to assess whether these approaches are complementary. This study concerns only floodplains of the Dutch parts of the Rhine and the Meuse. Causal relationships between species richness and the presence of policy target species are beyond our scope.

2 Material and methods

2.1 Biodiversity data

The data concerned ten rehabilitated floodplains along the rivers Meuse and Rhine (with its branches IJssel, Nederrijn and Waal) in the Netherlands (Figure 1 & Table 1).

Rehabilitation of the sites was realised by conversion from agricultural practice or conventional nature management to a strategy that included river dynamics and low-density grazing by horses and cattle as the main vegetation controlling processes. The sites were monitored from the first growing season after the conversion. Data concerning the situation before rehabilitation were not available.

The data may be described as follows:

1. Suitable data was available for six taxonomic groups; higher plants, birds, mammals, herpetofauna, odonates and butterflies.
2. Important characteristics of the ten sites are their size (acreage) and the river system they are part of (Table 1, Figure 1).
3. For each taxonomic group and site, the years of observation were known. The years of observation are ranging from 1989 to 1996.

The available data were used to calculate species richness (R, approach 1) and indices of the model BIO-SAFE (TBS, ***Taxonomic group Biodiversity Saturation*** index, approach 2) for each site per year per taxonomic group (see Figure 2). More sophisticated ecological methods were considered, but could not be applied due to limitations of the data set. An explanation of the model BIO-SAFE is given in De Nooij ***et al.*** (2004) and Lenders ***et al.*** (2001).

Figure 1. Geographical locations of ecological rehabilitation sites along large rivers in the Netherlands. 1. Eijsder Beemden; 2. Kleine Weerd; 3. Hochter Bampd; 4. Dilkensweerd; 5. De Horst; 6. Koningssteen; 7. Isabellegreend; 8. Ewijkse Plaat; 9. Millingerwaard; 10. Ossenwaard.

Table 1. The study sites and the data sources (the numbers of the sites correspond with Figure 1).

Nr.	Name of site	River	Acreage (ha)	Year of conversion	Data source
1	Eijsder Beemden	Meuse	61	1994	Van der Coelen, 1995; Lejeune & Kurstjens, 1996
2	Kleine Weerd	Meuse	12	1994	Van der Coelen, 1995; Lejeune & Kurstjens, 1997
3	Hochter Bampd	Meuse	45	1992	Shepherd ***et al.***, 1993a; Shepherd & Kurstjens, 1994; Van der Coelen, 1995
4	Dilkensweerd	Meuse	10	1992	Kurstjens & Shepherd, 1994a; Kurstjens, 1996a
5	De Horst	Meuse	10	1993	Kurstjens & Shepherd, 1994b
6	Koningssteen	Meuse	300	1990	Shepherd ***et al.***, 1991; 1992; 1993b; Kurstjens, 1996b
7	Isabellegreend	Meuse	34	1994	Kurstjens & Shepherd, 1995
8	Ewijkse Plaat	Waal (Rhine)	32	1989	Helmer, 1990; Helmer ***et al.***, 1991; Bosman, 1992a; 1994; 1995a; Bosman & Van der Veen, 1996
9	Millingerwaard	Waal (Rhine)	10 – 700*	1991	Bosman, 1992b; Bosman ***et al.***, 1993; Bekhuis ***et al.***, 1995
10	Ossenwaard	IJssel (Rhine)	50	1993	Bosman, 1995b; 1996; 1997

: increased in time

2.2 Statistical procedures

Species richness (R) and the ***Taxonomic group Biodiversity Saturation*** index (TBS) were compared on the basis of trend analysis and correlation analysis (Figure 2). The statistical techniques applied are standard for Analysis of Covariance (ANCOVA), the dependent variable being R or TBS, taken as numerical variables, and the explanatory variables being numerical variables (TBS or R respectively, Acreage, Year) as well as factors (Site, River).

The data were modelled as a multi-way ANCOVA model, according to the Wilkinson-Rogers notation for models (Wilkinson & Rogers, 1973). Trends were analysed with the models R ~ Site + Year, and TBS ~ Site + Year, correlations with R ~ Site + TBS. In order to justify significance levels, we applied post hoc analysis, performed using weights for the records, depending on the taxonomic group (taxon). For trends: R ~ taxon*Area + taxon*Year, and TBS ~ taxon*Area + taxon*Year. For correlations: R ~ taxon*Area + taxon*TBS, and TBS ~ taxon*Area + taxon*R. If the contribution of taxon*Year (or taxon*TBS, or taxon*R) showed to be significant, a search was done for the largest subset of taxonomic groups, for which Year or TBS did not show a significant contribution. All the models above were analysed with comparisons to models without the last terms, e.g. R ~ Site + Year, to R ~ Site (in this last example the influence of Year on R is studied, ***i.e.*** trend analysis).

The influence of the size of the sites, and the river system they were part of, also has been considered for both R and TBS. Below, examples for R are given. The same procedure was followed for TBS.For trend analysis, this was analysed with the comparisons R ~ taxon * (Site + Year + Acreage) to R ~ taxon * (Year + Acreage), and R ~ taxon * (Site + Year + River) to R ~ taxon * (Year + River).

For the correlation, this was analysed with the comparisons R ~ taxon * (Site + TBS + Acreage) to R ~ taxon * (TBS + Acreage), and R ~ taxon * (Site + TBS + River) to R ~ taxon * (TBS + River). The threshold p-value for significance was set to 0.0083, by applying the Bonferroni-correction (Miller, 1966).

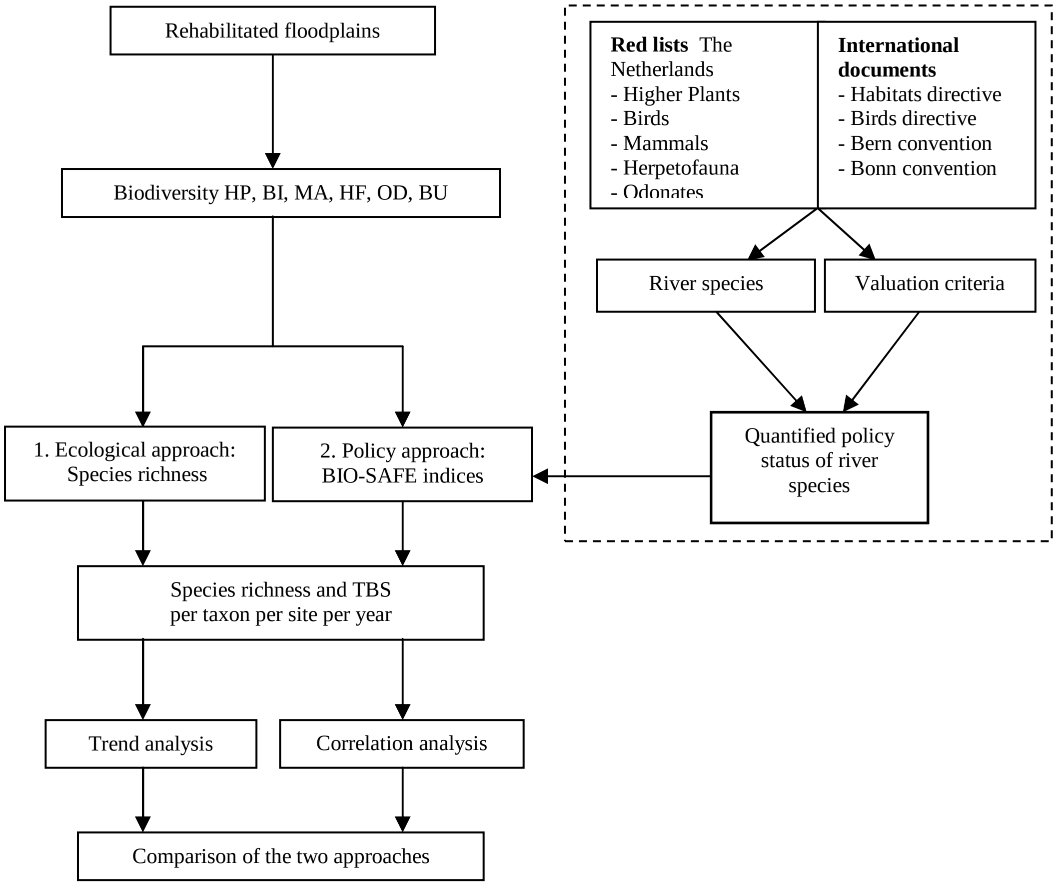

Figure 2. Protocol for biodiversity assessments and data analysis.
HP: Higher plants, BI: Birds, MA: Mammals, HF: Herpetofauna, OD: Odonates, BU: Butterflies, TBS: ***Taxonomic group Biodiversity Saturation*** index, Frame: Development of BIO-SAFE.

3 Results

Table 2 presents the outcomes of the ANCOVA. P-values always refer to the examined explanatory variable (EEV).

Table 2. Trends (analyses 1 and 2) and correlation (analysis 3) of R and TBS.

Analysis	Response	GEV	EEV	HP	BI	MA	HF	OD	BU
1	R	Site	Year	0,007*	0,056	0,030	0,004*	0,002*	0,013
2	TBS	Site	Year	0,044	0,240	0,130	0,021	0,075	0,036
3	R	Site	TBS	0,011	6,36 e-05*	0,146	3 e-04*	2,1 e-04*	0,131

*: significant, $p < 0.0083$ (cf. Bonferroni-correction), GEV: Given Explanatory Variable, EEV: Examined Explanatory Variable.

Table 2 shows p-values lower than 0.05 with respect to temporal trends of species richness of all groups except birds (analysis 1). However, only trends of higher plants, herpetofauna and odonates were significant. Trends of policy and legislation based indices exhibited p-values lower than 0.05 for higher plants, herpetofauna and butterflies (analysis 2). Following the Bonferroni-correction, none of these trends are significant. Significant correlations (analysis

3) between species richness and policy based values were found for birds, herpetofauna and odonates. For higher plants the p-value was low (0.011), but not significant.

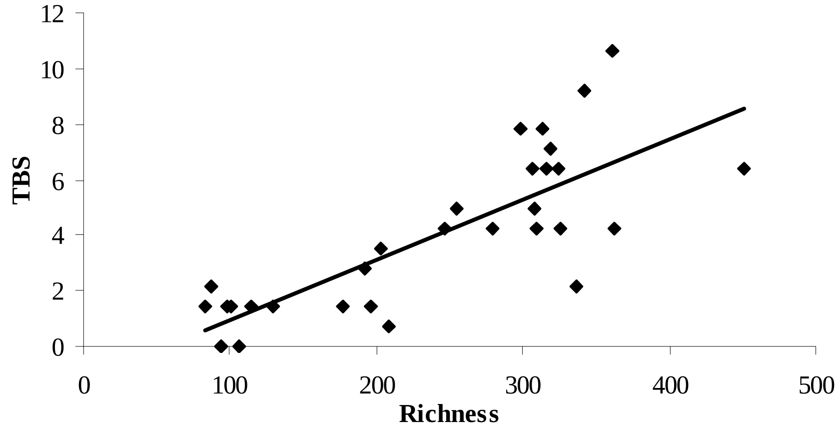

Figure 3. Correlation of species richness and TBS of higher plants.

The correlation of R and TBS for higher plants is shown in Figure 3. Although this correlation seems very strong, further analysis shows how the correlation can be dependent on the factor Site. Inclusion of the factor Site reveals positive as well as negative correlations (Figure 4). Concerning both trend analysis and correlation analysis, the hypothesis that Acreage or River system had an influence could be rejected ($p < 1e\text{-}15$, $p < 1e\text{-}10$, respectively).

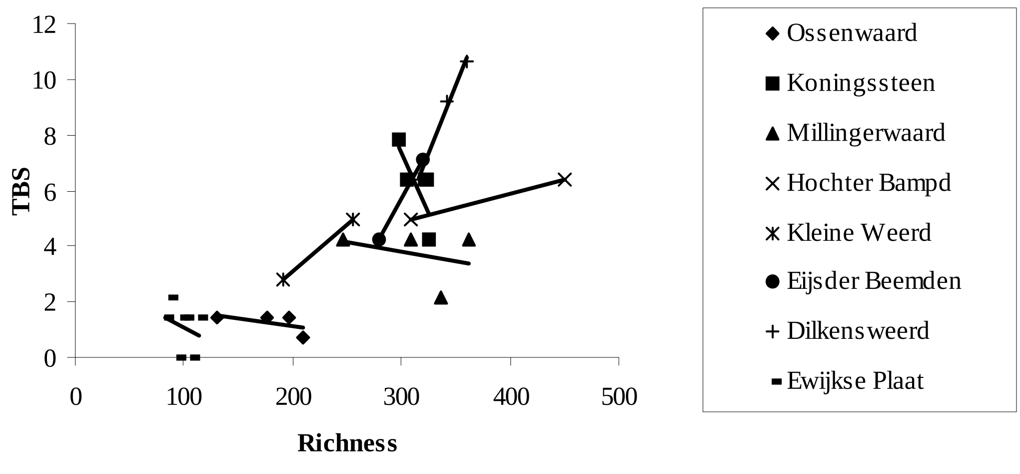

Figure 4. Correlation of R and TBS of higher plants for each site.

4 Discussion

This study concerned only rehabilitated floodplains of large lowland rivers in the Netherlands. These floodplains represent a very specific type of ecosystem and the results do not necessarily apply to other areas. For instance, the type of floodplain and the biogeography of the Dutch river systems might be important determinants for the biodiversity patterns we observed. Furthermore, species richness and our index based on protected and endangered species represent only two of the many diversity indices. The goal of this study was comparison of an ecological evaluation approach to a policy-based evaluation approach and to show some of the implications of this comparison. Species richness (approach 1) was compared with a policy and legislation based index, calculated by means of BIO-SAFE (approach 2). BIO-SAFE is a valuation model that incorporates only protected and endangered river species, derived from Red Lists, the European Habitats directive, the European Birds directive and the conventions of Bern and Bonn (De Nooij ***et al.***, 2004; Lenders ***et al.***, 2001).

4.1 Complementarity

Comparison of the two approaches for biodiversity assessment shows that the outcome of the evaluation differs per approach and taxonomic group. Both approaches yield interesting ***complementary*** information. Application of both methodologies therefore results in a more ***complete*** evaluation. The results show positive temporal trends with respect to species richness of higher plants, herpetofauna and odonates. The policy and legislation based index (BIO-SAFE, Lenders ***et al.***, 2001; De Nooij ***et al.***, 2004) increased for none of the species groups. Correlations between species richness and the BIO-SAFE index were found in all sites for birds, herpetofauna and odonates, and, to a lesser degree, higher plants.

The available data did not include the situation before rehabilitation and concerned periods of less than five years. Conclusions regarding the effects of the measures have to be regarded as preliminary. However, our results indicate that the floodplain rehabilitation measures were partly successful, and that there are various opportunities for improvement. For example, if species richness (R) increases but the policy based index (TBS) does not, then the point can be made that the measures as such are adequate, but need to be optimised in order to enhance possibilities for policy target species. This was the case for higher plants, herpetofauna and odonates. For higher plants, relations between R and TBS were positive as well as negative, depending on the site. Here R and TBS showed site-specific complementarity.

Conversely, when an evaluation indicates that neither species richness nor a policy based index increases, as was the case with birds, mammals and butterflies, then the recommendation to reconsider rehabilitation has a much stronger foundation. An evaluation focussed on species richness or policy target species alone would be less complete and overlook these points.

4.2 Surrogacy

The correlations found between species richness and policy based values were insensitive to the year of observation, the size of the site or the river system. Note that a correlation here is not taken as a causal explanation, but only applied as a means to compare two variables. However, if species richness and policy based indices are strongly correlated (as was the case with birds, herpetofauna and odonates), one might use the two approaches as surrogates for each other. Lund (2002) found that species of the Habitats directive can be used as indicators of species richness in Denmark. Oertli ***et al.*** (2002) examined the relation between pond area and biodiversity, and also found congruence between species richness and values based on Red List species. However, for the Dutch riverine landscape, more extensive research will be required to draw such conclusions. In view of these results and the upcoming measures for floodplain reconstruction (Smits ***et al.***, 2000), the possibility of using policy target species as indicators for biodiversity might be investigated in order to provide instruments for effective monitoring and impact assessment.

4.3 Requirements for sound assessments

Ecological rehabilitation has goals that are wider than mere increase of numbers of (protected and endangered) species. Comprehensive assessment therefore should also take the abiotic features of an ecosystem into account. However, assessing biodiversity is a primary step in prioritising conservation efforts, evaluation of rehabilitation measures and impact assessment. Sound assessments require consistent and complete data sets (including the situation before rehabilitation) that allow application of a variety of methodologies for assessment of ecological diversity. This variety can reflect the various aspects (e.g. taxonomic levels, spatial scale, heterogeneity and abundance, see Magurran, 1988) and dimensions (e.g. ecological, political and socio-economical) of diversity.

It is strongly recommended to use multiple biodiversity indicator systems (e.g. Noss, 1990) for monitoring and evaluation of rehabilitation projects. Such systems will allow consideration of the various aspects and dimensions of biodiversity mentioned above in the decision making process. This will put considerable demands on the quality of data and therefore on monitoring efforts. In any case, the choice of measured variables and their meaning in terms of the desired situation must be made explicit before starting monitoring and evaluation. Failure to do so will contribute to confusion and inconsistent management (Failing & Gregory, 2003).

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Chapter IV

Validity and sensitivity analysis of BIO-SAFE

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Abstract

Environmental Impact Assessment (EIA) must account for legally protected and endangered species. Uncertainties relating to the validity and sensitivity of EIA arise from predictions and valuation of effects on these species. This paper presents a validity and sensitivity analysis of a model (BIO-SAFE) for assessment of impacts of land use changes and physical reconstruction measures on legally protected and endangered river species. The assessment is based on links between species (higher plants, birds, mammals, reptiles and amphibians, butterflies and dragon- and damselflies) and ecotopes, and on value assignment to protected and endangered species using different valuation criteria (i.e. EU Habitats and Birds directive, Conventions of Bern and Bonn and Red Lists).

The validity of BIO-SAFE has been tested by comparing predicted effects of landscape changes on the diversity of protected and endangered species with observed changes in biodiversity in five reconstructed floodplains. The sensitivity of BIO-SAFE to value assignment has been analysed using data of a Strategic Environmental Assessment concerning the Spatial Planning Key Decision for reconstruction of the Dutch floodplains of the river Rhine, aimed at flood defence and ecological rehabilitation. The weights given to the valuation criteria for protected and endangered species were varied and the effects on ranking of alternatives were quantified.

A statistically significant correlation ($p < 0.01$) between predicted and observed values for protected and endangered species was found. The sensitivity of the model to value assignment proved to be low. Comparison of five realistic valuation options showed that different rankings of scenarios predominantly occur when valuation criteria are left out of the assessment. Based on these results we conclude that linking species to ecotopes can be used for adequate impact assessments. Quantification of sensitivity of impact assessment to value assignment shows that a model like BIO-SAFE is relatively insensitive to assignment of values to different policy and legislation based criteria. Arbitrariness of the value assignment therefore has a very limited effect on assessment outcomes. However, the decision to include valuation criteria or not is very important.

1 Introduction

1.1 Nature conservation in impact assessment

The goal of legal instruments for nature conservation is the conservation and, where appropriate, the rehabilitation of biodiversity. The Convention on Biological Diversity defines biodiversity as having three levels (genetic, species and ecosystem; UNEP, 1992). Noss (1990) looks at biodiversity from a different angle and distinguishes three components (compositional, structural and functional diversity). Each of the three levels of diversity can characterised and described using these three components (Le Maitre and Gelderblom, 1998). Composition then refers to presence of species, ecosystems and genes. Structure describes how elements of biodiversity are organised in space and time. Functional diversity refers to processes structuring ecosystems, populations and gene pools (Slootweg and Kolhoff, 2003). Legal instruments predominantly focus on the species level and the composition component of biodiversity. Furthermore, legislation only applies to a relatively small set of species, which are often endangered (European Environment Agency, 2005).

European legislation requires that impacts of physical planning and management on protected species are taken into account in Environmental Impact Assessment (EIA) and Strategic Environmental Assessment (SEA). In Europe, the legislative framework for nature conservation is formed by the Habitats directive (Council directive 92/43/EEC) and the Birds directive (Council directive 79/409/EEC). More in the background, the conventions of Bern, Bonn and Ramsar play a role in nature conservation policy. In the European Union, the Habitats and the Birds directive are considered to cover these conventions. Nature conservation legislation demands that effects of human activities are thoroughly assessed, weighed against other interests, and compensated for if necessary. The directives and conventions, as well as national legislation, make legal species protection a major issue in impact assessment. Another important policy and management instrument is formed by the Red Lists. Red Lists classify species into different categories that represent a certain extinction risk (IUCN, 2001). Red Lists are, compared to legal instruments, more science-based and give a better insight into which species have a higher chance of becoming extinct. Therefore, red-listed species are frequently used in addition to protected species in environmental impact assessments (Slootweg & Kolhoff, 2003).

1.2 Effect prediction and valuation

Prediction of effects of human activities and making choices between different project alternatives (e.g. in EIA and SEA) requires knowledge on species and their habitat demands. At initial phases of planning processes, detailed data on species abundance and their habitats is often lacking. Therefore, models that can predict and valuate potential presence of protected and endangered species based on simple data on species and landscape ecological units (cf. Lenders ***et al.***, 1998) are important in impact assessments.

Landscape ecological units are frequently described as so-called ecotopes (Klijn and Udo de Haes, 1994; Van der Molen ***et al.***, 2003). Ecotopes are spatial units of a certain extent, which are homogeneous regarding vegetation structure, succession stage and the main abiotic site factors that are relevant to plant growth (Klijn & Udo de Haes, 1994). Ecotopes are used by engineers, as well as by landscape ecologists and landscape designers. River ecotopes can relatively easy be mapped using remotely sensed data and combining digital terrain maps with hydrographs (Leuven ***et al.***, 2002; Van der Molen ***et al.***, 2003). This makes the concept of ecotopes a suitable tool for establishing input-output relations between models used in different disciplines applied in planning and management.

Apart from linking species to landscape units, impact assessment involves value assignments (Connelly and Richardson, 2005). Value assignments to protected and endangered species can reflect the difference between hard law (Habitats and Birds directive), soft law (the Bern, Bonn and Ramsar conventions) and policy and management instruments like Red Lists. Lists of protected species are often based on Red Lists. Therefore, lists concerning protected and endangered species usually partly overlap. A valuation approach could, for example, assign the highest values to endangered species protected by hard law, and the lowest values to species protected by soft law that are not endangered.

1.3 Uncertainty in effect prediction and valuation

Geneletti ***et al.*** (2003) explore and discuss the main uncertainties that are related to the typical stages of a biodiversity impact assessment: uncertainty in the data that are used (e.g. species and ecotope distribution), in the methodologies that are applied and in the value judgements provided by experts. Linkage of protected and endangered species to ecotopes and value assignment to protected and endangered species are two stages in environmental impact assessments from which uncertainty may arise.

The use of ecotopes for assessing impacts is an efficient method to quantify habitat potential and link species to patterns and processes of the physical environment. Harper (1995) distinguishes between potential habitats (main (a)biotic conditions are suitable as habitat) and functional habitats (the species actually lives there). The validity of using ecotopes (potential habitats) for impact assessment can be questioned (Corry & Iverson Nassauer, 2005). Linking species to ecotopes can not account for all causal factors that determine the potential value of a landscape to species and populations. Especially for animal groups, ecotopes have severe limitations when applied to describe their habitats. Ecotopes are a simplification of a species' habitat, required for modelling activities and visualisation. In this study, the validity of applying information about the presence and surface area of ecotopes to assess impacts of landscape changes is analysed.

In policy and management, species with a higher level of protection and/or extinction risk are given higher weights. Ultimately, weights given to individual species and their habitats determine the ranking of different alternatives in EIA. When protected and endangered species are concerned, value assignment can be based on the political-legal status of species. However, value assignment always remains somewhat arbitrary. It is therefore important to know to what extent the choice for a particular alternative is influenced by assignment of values to species. This is defined and analysed in this paper as sensitivity.

1.4 Validity and sensitivity of impact assessment concerning river floodplains

The abovementioned uncertainties relating to the validity and sensitivity of SEA and EIA are analysed in this paper related to reconstruction of river floodplains. In north-western Europe reconstruction measures aiming at flood defence and ecological rehabilitation are currently planned and carried out. These measures will have serious consequences for protected and endangered species in river-floodplain ecosystems. In the Netherlands, the plans include measures such as large scale floodplain excavation, reopening of secondary channels, river dike repositioning, removal of elevated areas and riverbed lowering. A comprehensive spatial planning project called 'Room for the River', which includes these measures, is currently being prepared (Van Stokkom ***et al.***, 2005).

A SEA concerning the Spatial Planning Key Decision 'Room for the River' for reconstruction of the Dutch floodplains of the river Rhine, aimed at flood defence and ecological rehabilitation, was carried out. In order to assess the consequences for protected and endangered riverine species, the model BIO-SAFE (Spreadsheet Application For Evaluation of BIOdiversity) was applied (Dutch Ministry of Transport, Public Works and Water Management, 2005). This model quantifies the importance of river ecotopes to protected and endangered river species (Lenders ***et al.***, 2001; De Nooij ***et al.***, 2004) based on the species' habitat demands and their political-legal status quantified via value assignment by an expert panel.

The objectives of this paper are to analyse the validity and sensitivity of impact assessment with BIO-SAFE. The following research questions are addressed:

1. How valid are assessments of impacts of physical reconstruction on protected and endangered species in river floodplains, using species-ecotope relations and changes in acreages of ecotopes?
2. What is the influence of value assignment to policy and legal instruments concerning protected and endangered species on the rankings of alternatives in impact assessments for river floodplain reconstruction?

2 Materials and methods

2.1 Model description

BIO-SAFE is a model for quantification of actual and potential values of protected and endangered species characteristic for river-floodplain ecosystems (Lenders ***et al.***, 2001, De Nooij ***et al.***, 2001, 2004). Based on their habitat demands, species are linked to river ecotopes. These links are used to quantify impacts on protected and endangered river species (Figure 1). The river ecotopes are classified using succession stage of vegetation, inundation frequency, morphodynamics and land use (Van der Molen ***et al.***, 2003). BIO-SAFE was developed for the floodplains of the rivers Rhine and the Meuse in the Netherlands, Germany, France and Belgium. In this study, however, only the Dutch version is analysed.

The information on species and habitats, required to select river species and to link species to ecotopes, was derived from a thorough survey of ecological literature, supplemented with expert knowledge. BIO-SAFE includes river species listed in the European Habitats Directive, the European Birds Directive, the Conventions of Bern and Bonn, the Red Lists (Figure 1, Table 1), and the Dutch Flora and Fauna Act. Taxonomic groups included in the model are higher plants, birds, herpetofauna, mammals, fish, butterflies and odonates. A panel of experts was sent a questionnaire in which they were asked to assign weights to the above-mentioned policy and legal instruments (valuation criteria), based on their importance to Dutch nature conservation policy. The panel distributed 40 points between the valuation criteria. Each species was assigned the summed weights of valuation criteria (Table 1) that mention the species. Applying the valuation criteria, leads to a quantified policy status score (***Species-specific*** score; S-score; Figure 1). For example, in De Nooij ***et al.*** (2004) to the White stork (which is mentioned in the Bern convention, Bonn convention and in the Birds Directive), a score of 21.4 was assigned. Through the linkage of species to ecotopes, values are assigned to ecotopes as well. On the basis of data on the surface area of ecotopes, BIO-SAFE calculates an aggregated score for different species groups on the floodplain level. This score represents the potential value of a floodplain regarding protected and endangered species. With these outputs alternatives can be ranked according to their impact on protected and endangered species (Figure 1).

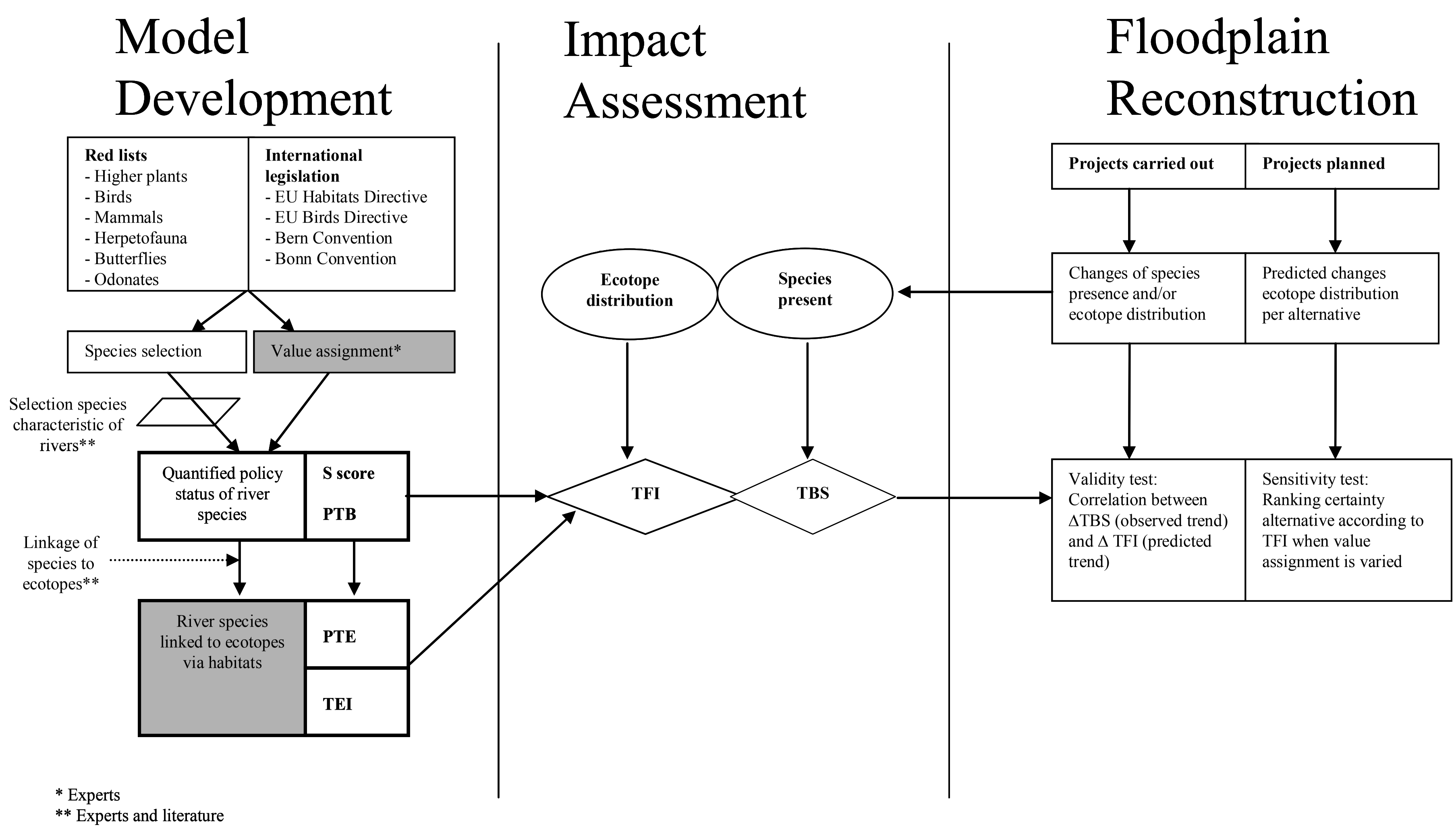

Figure 1. Coherence of development, application and analysis of validity and sensitivity of BIO-SAFE. Shaded boxes: sources of uncertainty investigated in this paper. Acronyms are explained in the text.

An explanation of the species selection process, value assignment, development of the ecotope typology and linkage of species to ecotopes, including a full description of the functionalities of the model BIO-SAFE can be found in Lenders ***et al.*** (2001) and De Nooij ***et al.*** (2001, 2004). The complementary value of evaluation of ecological rehabilitation on the basis of protected and endangered species in addition to evaluation with species richness and the relation between protected and endangered species and the hydrodynamic gradient were studied in De Nooij ***et al.*** (2005, 2006).

Table 1. Valuation criteria applied and their mean weights and standard deviation (between brackets) for the Netherlands according to a Dutch expert panel (n=17, De Nooij ***et al.***, 2004) and numbers of species meeting these criteria, per taxonomic group for the Dutch version of the model BIO-SAFE.

Taxon	Red Lists[1] (6.9 ± 3.2)	Habitats Directive[2] (11.6 ± 3.1)	Birds Directive[3] (10.1 ± 2.8)	Convention of Bern[4] (5.7 ± 2.4)	Convention of Bonn[5] (5.6 ± 2.7)	All criteria (40)
Higher plants	136	2	n.a.	2	n.a.	136
Birds	27	n.a.	22	37	47	60
Herpetofauna	6	9	n.a.	5	0	9
Mammals	6	6	n.a.	5	3	9
Fish	19	11	n.a.	1	1	20
Butterflies	17	2	n.a.	2	0	17
Odonates	6	3	n.a.	3	0	6
All groups	217	33	22	55	51	257

1: IUCN-criteria ‘extinct’, ‘critical’, ‘endangered’ and ‘vulnerable’ or ‘susceptible’.
2: Annex II: species whose conservation requires the designation of special areas of conservation; Annex IV: species in need of strict protection; Annex V: species whose taking in the wild and exploitation may be subject to management measures.
3: Annex I: species that are subject of special conservation measures concerning their habitat in order to ensure their survival and reproduction in their area of distribution.
4: Appendices I and II: strictly protected flora and fauna species respectively.
5: Appendix I: migratory species whose immediate protection is required; Appendix II: migratory species whose conservation and management should be covered by means of transnational agreements.
n.a.: not applicable (EU-Birds Directive and EU-Habitats Directive are complementary. EU-Birds Directive applicable to birds only; EU-Habitats Directive applicable to all other species).

The S-scores of species belonging to a particular taxonomic group are summed to yield a ***Potential Taxonomic group Biodiversity*** (PTB) constant. This score reflects the maximum score possible for the taxonomic group involved.

For each ecotope type, the S-scores assigned to the species linked to the ecotope are summed per taxonomic group, yielding a ***Potential Taxonomic group Ecotope*** (PTE) constant (Figure 1), i.e. the maximum score for an ecotope regarding a particular taxonomic group. Subsequently, this PTE constant is related to the PTB constant, resulting in a ***Taxonomic group Ecotope Importance*** score (TEI), ranging from 0 to 100 per ecotope type (Figure 1; equation 1).

$$TEI = PTE * 100 / PTB \qquad [1]$$

TEI = ***Taxonomic group Ecotope Importance*** score
PTE = ***Potential Taxonomic group Ecotope*** constant
PTB = ***Potential Taxonomic group Biodiversity*** constant

Data on presence of species in the areas studied are used to calculate the biodiversity saturation of an area per taxonomic group (TBS; equation 2).

$$TBS = ATB * 100 / PTB \qquad [2]$$

TBS = ***Taxonomic group Biodiversity Saturation*** index
ATB = ***Actual Taxonomic group Biodiversity*** score
PTB = ***Potential Taxonomic group Biodiversity*** constant

Assessment of impacts of different reconstruction alternatives for each taxonomic group are calculated by multiplying the ***Taxonomic group Ecotope Importance*** scores per ecotope type by the relative surface area, derived from GIS maps, of that particular ecotope type to be realised within each alternative. This is done for each alternative. These products are summed per taxonomic group, thus offering insight into the significance of each alternative for that particular taxonomic group. This results in the ***Taxonomic group Floodplain Importance*** score (TFI; equation 3). The ***Taxonomic group Floodplain Importance*** score (TFI) of different taxonomic groups can be summed up per alternative yielding the ***Floodplain Importance*** score (FI; equation 4). This procedure means that when the surface area of an ecotope increases by 20% the importance of this ecotope will also increase by 20%. A further explanation of the calculation and use of the indices and scores is given in De Nooij ***et al.*** (2001).

$$TFI_y = \Sigma \left((SA_{ecotope(x)} / SA_{floodplain}) * TEI_x \right) \qquad [3]$$

TFI_y = ***Taxonomic group Floodplain Importance*** score for taxonomic group y
$SA_{ecotope(x)}$ = surface area of ecotope x present in the floodplain
$SA_{floodplain}$ = surface area of the floodplain
TEI_x = ***Taxonomic group Ecotope Importance*** score for ecotope type x

$$FI = \Sigma\, TFI_y \qquad [4]$$

FI = ***Floodplain Importance*** score
TFI_y = ***Taxonomic group Floodplain Importance*** score for taxonomic group y

2.2 Validity analysis

Study area and data sources

Data for evaluation of validity of the model concerned five floodplains along the river Rhine (Table 2; geographical locations in Figure 2a). These areas were reconstructed with the purpose of flood defence and ecological rehabilitation. Examples of measures are floodplain lowering by clay excavation and in some cases the reopening of secondary channels and removal of summer dikes. Before reconstruction, the land-use was mainly agricultural. After reconstruction the areas were managed following a strategy that includes influence of river dynamics and low-density grazing by horses and cattle as the main processes controlling vegetation succession (Pelsma, 2002; Aggenbach and Pelsma, 2005). Data concerning species presence before and after reconstruction were derived from various monitoring reports and NGO databases (Table 2). Ecotope data were available in GIS vector format. Surface areas of ecotopes before and after reconstruction were calculated using ArcView GIS Version 3.1. The ecotopes were based on an ecotope typology composed of 60 ecotopes, suitable for maps with a scale of 1:10 000. Data for fish were not sufficiently available for all five areas.

Table 2. Reconstructed floodplains and data sources (numbers refer to Figure 2a).

Nr	Name of site	Rhine branch	Year of conversion	Species data	Ecotope data
1	Engelse werk (30 ha)	IJssel	1992	1993[g], 1997[g]	1993[f], 1997[f]
2	Duursche waarden (120 ha)	IJssel	1990	1992[a,d,e], 1996[a,d,e]	1991[b], 1993[b], 1996[b]
3	Blauwe kamer (176 ha)	Nederrijn	1992	1993[a], 1994[a], 2000[a]	1993[b], 1994[b], 2000[b]
4	Afferdensche en Deestsche waarden (300 ha)	Waal	1996	1995[a], 1997[a], 2000[a]	1995[b], 1997[b], 2000[b]
5	Ewijkse plaat (32 ha)	Waal	1989	1989[c], 1991[c], 1995[c]	1989[c], 1991[c], 1995[c]

[a] Data obtained from NGO's collecting and managing flora and fauna data in the Netherlands, [b] Data obtained from Ministry of Transport, Public works and Water Management [c] Drouet (2002), [d] De Goeij ***et al.*** (1998), [e] Verbeek ***et al.*** (1998), [f] Data obtained from Province of Overijssel (the Netherlands), [g] Bremer (1998)

Data processing and statistics

The data concerning reconstructed floodplains were used to calculate TBS and TFI values for the situation before and after landscape changes, on the basis of protected and endangered species presence and ecotope abundance respectively (Figure 1). For each area (n=5) and taxonomic group (n=6) ΔTBS and ΔTFI values were calculated (equation 5 and 6). The ΔTFI values were expressed in percentage change between before and after reconstruction, the ΔTBS values already were expressed in percentage.

$$(\Delta TBS)_{t1,t0} = TBS_{t1} - TBS_{t0} \quad [5]$$

$$(\Delta TFI)_{t1,t0} = (TFI_{t1} - TFI_{t0}) / TFI_{t0} * 100\% \quad [6]$$

TBS = ***Taxonomic group Biodiversity Saturation*** index
TFI = ***Taxonomic group Floodplain Importance*** score
t0 = before reconstruction, t1 = after reconstruction

Subsequently, all ΔTBS and ΔTFI values were ranked (n=24). Some samples were omitted due to poor data quality. By means of SPSS 10.0, a Pearson correlation was carried out to determine if rankings of ΔTBS (observed changes in value of protected and endangered species present) correlate significantly with ΔTFI (predicted changes in the importance of floodplains studied for protected and endangered species). In this regression analysis all taxonomic groups and areas were taken together as one dataset because the data available did not allow statistical testing per group and area.

2.3 Sensitivity analysis

Study area and data sources

The study area for the sensitivity analysis concerned most of the floodplains along the Dutch branches of the river Rhine (Waal, Nederrijn and IJssel, Figure 2b). Only the estuarine zones were excluded. The data used for the sensitivity analysis concerned 13 river segments. For each river segment, four alternatives (Table 3) for flood defence and ecological rehabilitation were available (Dutch Ministry of Transport, Public Works and Water Management, 2005):

1. Alternative 1 (A1), Autonomous development, in this alternative no measures aimed at reducing flooding risks are taken, but existing plans for nature development are carried out as planned.
2. Alternative 2 (A2), focussed on keeping the costs within budget, relatively conservative.
3. Alternative 3 (A3), focussed on spatial quality, relatively nature oriented.
4. Alternative 4 (A4), a combination of A2 and A3.

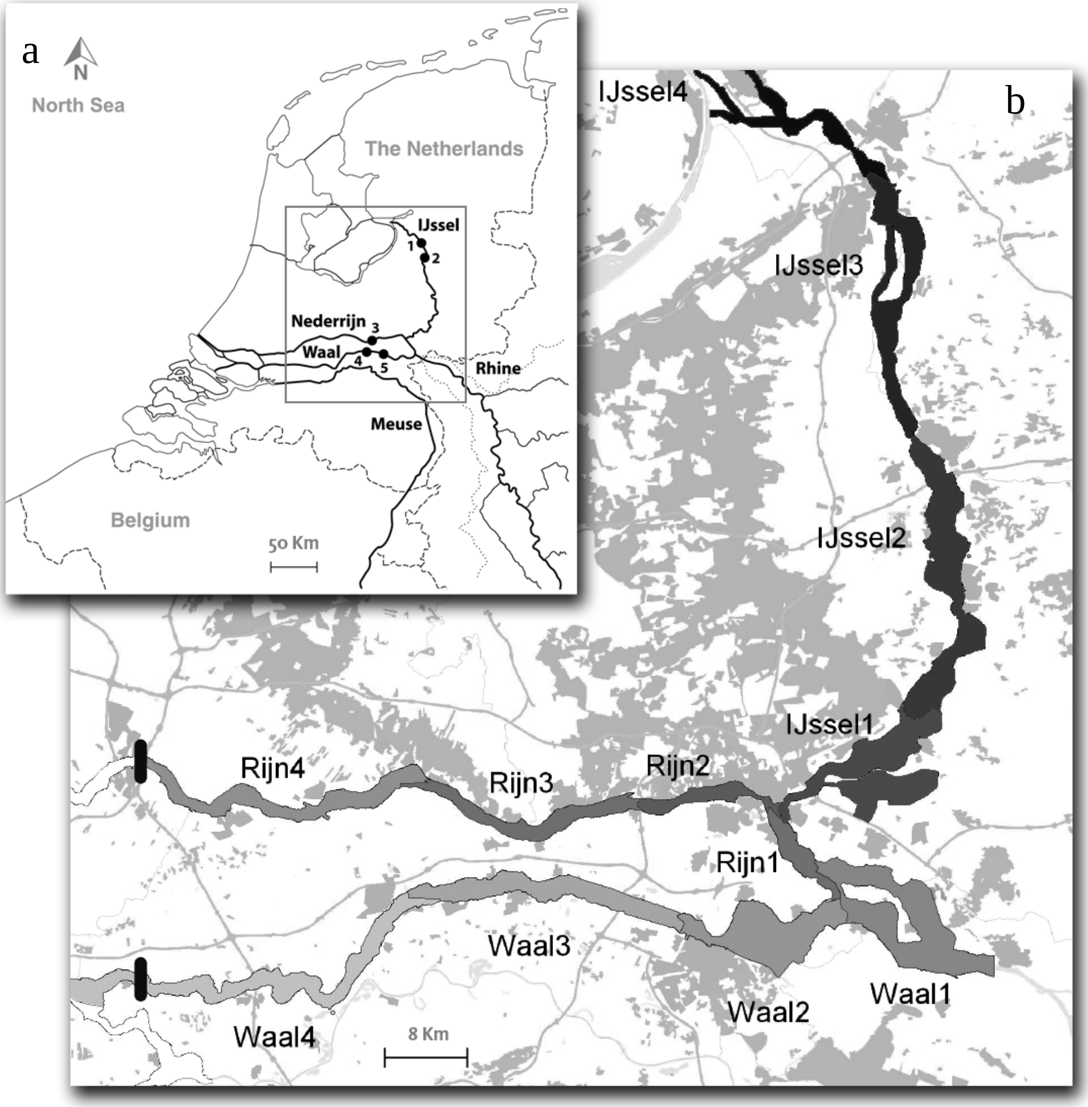

Figure 2. Study area a): Locations of reconstructed floodplains, 1. Engelse werk, 2. Duursche waarden, 3. Blauwe kamer, 4. Afferdensche en Deestsche waarden, 5. Ewijkse plaat; study area b): River reaches for which reconstruction plans are developed in the project 'Room for the River'. Rijn, Waal and IJssel are the distributaries of the river Rhine in the Netherlands.

In all four alternatives (Table 3) there is a large decrease of the surface area of man-made ecotopes like built-up area, production grassland and arable land compared to the present situation. More natural ecotopes like marsh, natural grassland, natural levee pasture and alluvial forest will increase. Large differences between the alternatives occur for the ecotope river dune and natural grasslands.

The ecotope data for each alternative were generated by means of hydromorphodynamic and vegetation/succession models and made available as data tables derived from GIS maps (mapping scale 1:50 000) by the Dutch Ministry of Transport, Public Works and Water Management. The impact assessment was not done for fish because the definition of aquatic ecotopes in the ecotope typology applied was judged too coarse for realistic fish habitat description.

Table 3. Expected landscape changes in 13 river segments according to four different alternatives, expressed in % relative to the present situation (Ministry of Transport, Public Works and Water Management, 2005).

Ecotope	A1	A2	A3	A4
Built-up area	-62	-62	-57	-57
Production grassland	-26	-26	-27	-27
Arable land	-23	-23	-23	-24
Marsh	+210	+210	+210	+203
Natural grassland	+324	+384	+337	+395
Natural levee pasture	+55	+53	+51	+40
Softwood alluvial forest	+18	+15	+19	+12
Hardwood alluvial forest	+16	+16	+15	+34
River dune	-30	-30	+45	+41

A1 = Alternative 1, A2 = Alternative 2, A3 = Alternative 3, A4 = Alternative 4

Data processing and statistics

The data concerning all 13 river reaches together were used to calculate rankings of alternatives (Figure 1) on an aggregated level of spatial scale. The analysis was also carried out on a lower spatial scale, i.e. that of separate river reaches. Two river reaches were selected (IJssel 2, Rijn 5) for the sensitivity analysis because preliminary analysis showed that the alternatives for these reaches gave relatively large differences with respect to their importance for taxonomic groups involved. The total size of the area on the aggregated level was 56 000 ha, for ‘IJssel 2’ 8 200 ha and for ‘Rijn 5’ 1 650 ha. For the reach Rijn 5, the combination alternative (A4) corresponded entirely with the autonomous development (A1).

The sensitivity of the ranking results to value assignment to different policy and legislation lists (Table 1) was evaluated using Monte Carlo simulation (Vose, 1996). This simulation technique propagates the uncertainty about assigned values into an uncertainty distribution of the ranking results. Each value assigned (Table 1) was replaced by a uniform distribution that varied between 0 and 40 to reflect the input uncertainty to reflect the possible uncertainty of the value assigned. Uniform distribution means that each possible value within the defined range (0-40) has an equal probability of getting selected. The sum of the simulated individual values was normalized to 40 to guarantee consistency with the weights allotted by the expert panel. Monte Carlo simulations were performed using the software package Crystal Ball (Decisioneering Inc., 1999a) in Latin Hypercube mode with 10 000 iterations, which is generally considered sufficient to obtain a representative frequency chart of the output variables (Morgan & Henrion, 1990). The Latin Hypercube mode ensures even sampling over the probability range of the distribution (Decisioneering Inc., 1999b).

Ranking certainty was studied for an aggregated level (i.e. all reaches together) as well as for the two selected river reaches for each taxonomic group and for all groups taken together (Table 4). The frequency of each ranking order as well as the frequency of each ranking position (1-4) of each alternative in the 10 000 samples created was calculated. This gives insight into the certainty of each possible ranking of alternatives and the certainty of a choice for a particular alternative. The higher the certainty, the lower the sensitivity of BIO-SAFE to value assignment is.

Apart from analysing the certainty of ranking of alternatives, different valuation options were compared. From all possible value distributions, a few realistic combinations were selected. With BIO-SAFE, TFI values for each alternative were calculated according to the following valuation options:

1. All protected and endangered species are taken into account, value assignment according to De Nooij ***et al.*** (2004).

2. All protected and endangered species are taken into account, uniform value assignment.
3. Only species protected by EU legislation are taken into account.
4. Only endangered species (red-listed species) are taken into account.
5. Only nationally protected (Dutch Flora and Fauna Act) species are taken into account.

Different valuation options were compared by means of Spearman Rank correlation to determine if significant differences between valuation options occur. This was done for each taxonomic group and all groups taken together on the aggregated level as well as on the level of the two selected river reaches.

3 Results

3.1 Validity analysis

Figure 3 shows the rankings of changes in TBS and TFI. Pearson correlation showed that when changes in TBS and TFI are ranked, these rankings correlate significantly (n=24, R^2=0.54, $p < 0.01$, 1-tailed) following:

$$r(\Delta TBS)_{t,g} = 5,8 + 0.54\ r(\Delta TFI)_{t,g}$$

$r(\Delta TBS)_{t,g}$: rank of ΔTBS values for all areas and taxonomic groups
$r(\Delta TFI)_{t,g}$: rank of ΔTFI values for all areas and taxonomic groups

This result means that there is a significant positive correlation between effects of landscape changes on the predicted potentials for protected and endangered species and observed changes in actual values for protected and endangered species quantified by BIO-SAFE.

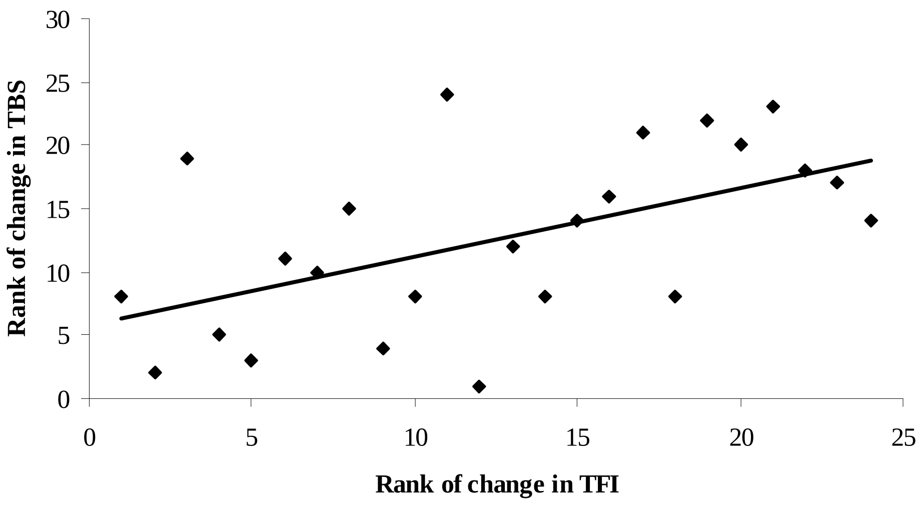

Figure 3. Correlation between rankings of ΔTFI (predicted biodiversity change) and ΔTBS (observed biodiversity change). Acronyms are explained in the text.

3.2 Sensitivity analysis

Certainty of ranking of reconstruction alternatives

Table 4 presents the results of the sensitivity analysis with Crystal Ball. The certainties of the most probable ranking of alternatives, and the certainties of the position of each alternative are given. Certainties are presented for all taxonomic groups separately, as well as taken together, on an aggregated level and on the level of two separate river reaches.

On the aggregated level, ranking of alternatives for all taxonomic groups have a certainty above 95% except for butterflies (94.8%). Rankings for higher plants have a certainty of 98.4%. For the other taxonomic groups, as well as for all groups taken together, the certainty is 100%. Alternative A3 ranks highest for most separate groups, as well as for all groups taken together. Only for higher plants, alternative A3 ranks second behind alternative A4. The certainty of the number 1 position of alternative A3 is 100%, except for butterflies (94.8%). The certainty of the number 1 position of alternative A4 for higher plants is 99.7%. When the rankings of the different alternatives are compared for different taxonomic groups, it shows that for most groups the rankings are identical. Only higher plants and mammals deviate from this general pattern.

Table 4. Certainty of rankings and positions of alternatives in the rankings. A high certainty corresponds with a low sensitivity.

Study area and taxon	Alternatives ranking position and certainty percentage (between brackets)				Total rank certainty percentage
All reaches	A1	A2	A3	A4	
Higher plants	3 (98,4)	4 (98,7)	2 (98,7)	1 (99,7)	98,7
Birds	4 (100)	3 (100)	1 (100)	2 (100)	100
Mammals	3 (100)	4 (100)	1 (100)	2 (100)	100
Herpetofauna	4 (100)	3 (100)	1 (100)	2 (100)	100
Butterflies	4 (100)	3 (100)	1 (94,8)	2 (94,8)	94,8
Odonates	4 (100)	3 (100)	1 (100)	2 (100)	100
All groups	4 (100)	3 (100)	1 (100)	2 (100)	100
IJssel 2	A1	A2	A3	A4	
Higher plants	2 (99,7)	3 (99,8)	4 (100)	1 (99,7)	99,7
Birds	3 (100)	4 (100)	2 (100)	1 (100)	100
Mammals	3 (100)	4 (100)	2 (86,9)	1 (86,9)	86,9
Herpetofauna	3 (100)	4 (100)	2 (100)	1 (100)	100
Butterflies	2 (93,0)	3 (88,3)	4 (88,3)	1 (100)	88,3
Odonates	3 (100)	4 (100)	2 (100)	1 (100)	100
All groups	3 (100)	4 (100)	2 (100)	1 (100)	100
Rijn 5	A1/A4	A2	A3		
Higher plants	1 (100)	2 (93,6)	3 (93,6)		93,6
Birds	1 (100)	3 (100)	2 (100)		100
Mammals	1 (100)	3 (100)	2 (100)		100
Herpetofauna	1 (100)	3 (100)	2 (100)		100
Butterflies	3 (67,7)	1 (92,5)	2 (60,3)		60,3
Odonates	1 (100)	3 (100)	2 (100)		100
All groups	1 (100)	3 (100)	2 (100)		100

A1 = Alternative 1, A2 = Alternative 2, A3 = Alternative 3, A4 = Alternative 4

On the reach level, for the reach IJssel 2, the certainty of ranking of alternatives is above 95%, except for mammals and butterflies. Ranking of alternatives for higher plants have a certainty of 99.7%. Rankings for birds, herpetofauna and odonates, as well as for all groups taken together, have certainties of 100%. Alternative A4 ranks highest for all groups, and all groups taken together. The certainty of this position is 100%, except for higher plants (99.7%) and mammals (86.9%). For the reach Rijn 5, the certainty of ranking of alternatives is above 95%, except for higher plants (93.6%) and butterflies (60.3%). Ranking of alternatives for birds, mammals, herpetofauna and odonates, as well as for all groups taken together, have certainties of 100%. Alternative A1/A4 ranks highest for all groups separately and all groups taken together (certainty 100%), except for butterflies where this alternative has the lowest rank (certainty 67.7%). On the reach level, rankings according to scores of alternatives for higher plants and butterflies differ from rankings according to the other taxa for both IJssel 2 and Rijn 5.

The results presented in Table 4 mean that indices for higher plants, butterflies and mammals are slightly sensitive to value assignment. Indices for birds, herpetofauna, odonates and all groups together, show no sensitivity at all.

Comparison of different valuation options

Comparison of different valuation options by means of Spearman rank correlation shows that all valuation options give identical rankings of alternatives, when rankings are based on indices for all groups taken together (Table 5). For the individual taxonomic groups, there are no differences in rankings between option 1 and option 2 (all criteria included, valuation according to an expert panel vs. all criteria included, all criteria weighed equal). Valuation options that are also identical in ranking of alternatives are option 1 and 5 (all criteria included, valuation according to an expert panel vs. only species protected according to the Dutch Flora and Fauna act) and option 2 and 5 (all criteria included, all criteria weighed equal vs. only species protected according to the Dutch Flora and Fauna act). This applies to the aggregated level as well as the reach level.

On the aggregated level, differences in ranking occur for some species groups when criteria are left out. For higher plants, valuation on the basis of EU legislation only (option 3) always gives different rankings compared to the other valuation options. For mammals and butterflies, valuation on the basis of Red Lists only (option 4) always gives different rankings compared to other options.

On the reach level the pattern is largely similar to that of the aggregated level, with some additional results. Option 3 (valuation on the basis of EU-legislation only) also differed significantly from all other options for the groups mammals and butterflies for the reach IJssel 2. Furthermore, option 3 differed significantly from option 1 (all criteria included, valuation according to an expert panel) and option 2 (all criteria included, all criteria weighed equal) for butterflies for the reach Rijn 5.

The results of Table 5 show that different rankings of reconstruction alternatives occur for plants, mammals and butterflies, when only EU Habitats and Birds directive (international hard law criteria) are applied. For mammals and butterflies, applying only the Red list criterion also leads to different rankings.

Table 5: Taxonomic groups with significant differences ($p < 0.05$) between rankings according to different valuation options.

Study area and taxon	Comparison of valuation options									
	1-2	1-3	1-4	1-5	2-3	2-4	2-5	3-4	3-5	4-5
All reaches										
Higher plants	-	*	-	-	*	-	-	*	*	-
Birds	-	-	-	-	-	-	-	-	-	-
Mammals	-	-	*	-	-	*	-	*	-	*
Herpetofauna	-	-	-	-	-	-	-	-	-	-
Butterflies	-	-	*	-	-	*	-	*	-	*
Odonates	-	-	-	-	-	-	-	-	-	-
All groups	-	-	-	-	-	-	-	-	-	-
IJssel 2										
Higher plants	-	*	-	-	*	-	-	*	*	-
Birds	-	-	-	-	-	-	-	-	-	-
Mammals	-	*	*	-	*	*	-	*	*	*
Herpetofauna	-	-	-	-	-	-	-	-	-	-
Butterflies	-	*	-	-	*	-	-	*	*	-
Odonates	-	-	-	-	-	-	-	-	-	-
All groups	-	-	-	-	-	-	-	-	-	-
Rijn 5										
Higher plants	-	*	-	*	*	-	*	*	-	*
Birds	-	-	-	-	-	-	-	-	-	-
Mammals	-	-	-	-	-	-	-	-	-	-
Herpetofauna	-	-	-	-	-	-	-	-	-	-
Butterflies	-	*	*	*	*	*	*	-	-	-
Odonates	-	-	-	-	-	-	-	-	-	-
All groups	-	-	-	-	-	-	-	-	-	-

Option 1. All protected and endangered species are taken into account, value assignment according to De Nooij ***et al.*** (2004); Option 2. All protected and endangered species are taken into account, all criteria weighed equal; Option 3. Only species protected by EU Habitats and Birds directive are taken into account; Option 4. Only endangered species (red-listed species) are taken into account; Option 5. Only nationally protected (Dutch Flora and Fauna Act) river characteristic species are taken into account.
- = Identical rankings, *: Significantly different ranking ($p<0.05$).

4 Discussion

The objectives of this study were to investigate uncertainties relating to the validity and sensitivity of EIA, arising from the use of ecotopes as potential habitats, and value judgements, respectively (cf. Geneletti ***et al.***, 2003). This was done using a model for quantification of the impact of river-floodplain reconstruction measures on protected and endangered species (BIO-SAFE), which is based on linkage of species and ecotopes and value assignment to political and legal criteria by an expert panel.

4.1 BIO-SAFE methodology

BIO-SAFE is an operational model for integration of ecological knowledge with legal instruments in river management. It can assess actual and potential values of river floodplains and ecotopes in these areas, and reconstruction plans based on 257 protected and endangered river characteristic species and their habitats. BIO-SAFE can therefore be used as a tool for biodiversity assessment with regard to design and evaluation of physical planning projects, management measures, environmental impact assessments, and comparative landscape-ecological studies. The instrument can measure progress towards goals of nature policy and legislation.

BIO-SAFE is well adapted to the methods of river management in north-western Europe, which characterise river-floodplain ecosystems by means of ecotopes. Moreover, the model can already be used when only information on presence/absence of species and ecotopes is available. Species that are not protected and/or endangered are left out. Many protected species are not the species with the highest extinction risk and endangered species are not necessarily the most important ones for ecosystem functioning. However, De Nooij ***et al.*** (2006) show that the habitat demands of the 275 protected and endangered species in the model are such, that optimisation of potentials for these species will result in rehabilitation of conditions that are also required by the riverine species assemblage as a whole.

4.2 Validity of impact assessment

Changes in the importance of floodplains for protected and endangered species predicted on the basis of the observed ecotope situation before and after reconstruction correlate significantly ($p < 0.01$) with changes in values based on observed protected and endangered species presence in the floodplain before and after reconstruction. This means that the model BIO-SAFE can use ecotope data to adequately assess impacts of reconstruction measures on protected and endangered species. It can be concluded that linking species to ecotopes can be a useful and valid methodology in impact assessments. Ideally, however, an impact assessment model would include not just ecotopes, but all the information on species' autecology, (meta)population dynamics and life history. Ecotopes should be regarded as potential habitats (sensu Harper, 1995), and are a simplification of functional habitats of species. In this study, no information on the actual function of the ecotopes for protected and endangered species was included. Therefore, the validity of our method is a remarkable result.

The study presented here was based on short time periods (< 5 years), and a very limited number of floodplain areas (n=5). Therefore, we must be careful in concluding that the assessment of impacts using BIO-SAFE is valid in a generic sense. This statement would require more rigorous and extensive study (more areas with different ecotope distributions, longer time periods) that gives insight into the validity of the model for each species group separately and on larger time scales. However, such data are currently not available for reconstructed floodplains of the rivers Rhine and Meuse (Buijse ***et al.***, 2005).

4.3 Certainty of ranking of reconstruction alternatives

The ranking of reconstruction alternatives predominantly proved insensitive to value assignment. When the weights are varied randomly between 0 and 40, the certainty of the choice for a particular reconstruction alternative (i.e. the certainty of the number 1 position of a reconstruction alternative) is always above 85% and usually above 99%. When ranking of reconstruction alternatives is based on the score for all groups taken together, birds, herpetofauna or odonates, the certainty is always 100%. Taxonomic groups for which the certainty of ranking of alternatives often is less than 100% are higher plants and butterflies. For mammals this occurred only once. A relatively low certainty (60.3%) occurred for butterflies in one case.

The sensitivity, or insensitivity, of ranking of reconstruction alternatives based on BIO-SAFE indices for particular taxonomic groups can be explained by (combinations of) two factors:

1. The importance of particular valuation criteria for the different taxonomic groups (i.e. the political and legal status of species and the corresponding distribution of value within the taxonomic groups, and degree of overlap between the valuation criteria).
2. The variation of the importance of the different ecotopes (TEI values) within the taxonomic groups, related to the degree to which species are specific for one or few ecotopes.

The insensitivity of the indices for birds, herpetofauna and odonates can be explained by both factors. Most species of these groups feature on all lists that were used as valuation criteria (high overlap) and the TEI values show little variation within these groups, due to the fact that a lot of species of these groups are linked to many ecotopes. Species of the sensitive groups of higher plants and butterflies are relatively specific for certain ecotopes and there is a large amount of variation between the importances of different ecotopes. Furthermore, species of these two groups are mainly derived from the Red Lists only (low overlap; see Table 1). Therefore, value distribution within these taxonomic groups is highly uneven. For mammals, sensitivity is probably mainly caused by the value distribution within this species group. Further research must point out how the two above-mentioned factors interact and how much influence each factor has.

Certainty of ranking of reconstruction alternatives increases with higher levels of aggregation. This occurs both when reaches (higher spatial scale) and taxa (higher taxonomic level) are aggregated. Apparently this means that the resolution of the model decreases with increasing scale. This can be explained by the fact that on a higher spatial scale the ecotope data exhibit a lower degree of variability: the surface area is more evenly distributed over the ecotopes on the aggregated level than in the separate reaches selected. The reaches studied here were selected because of the large differences between the taxonomic groups, caused by large differences in surface areas of ecotopes (uneven distribution of surface area over the ecotopes). Variability of the ecotope importance scores and value distribution within each taxonomic group is largely lost with aggregation of the different groups.

These results mean that model outputs are relatively independent from value judgements concerning the status of protected and endangered species and therefore the arbitrariness of the value assignment (or the subjectivity of an expert panel) has a very limited effect. BIO-SAFE is a suitable tool even if there is no clear consensus on the relative weights that should be given to the various political and legal criteria.

4.4 Comparison of different valuation options

Comparison of five different realistic valuation options is used to derive conclusions about sensitivity for particular criteria. When rankings are based on indices for all groups taken together, no differences occur between the different valuation options. On the level of the taxonomic groups, there are no differences between an option that uses value assignment according to an expert panel and an option where all weights are equal, as long as all criteria are applied. However, when criteria are left out, this significantly influences outputs of BIO-SAFE. For plants, mammals and butterflies, different rankings of reconstruction alternatives occur when only EU Habitats and Birds directive (international hard law criteria) are applied.

This can be explained by the fact that the species selection for these groups consists predominantly of species that are only mentioned in Red Lists. The group of plants contains only 2 species listed in the Habitats directive out of 135 plant species in total. Only 2 butterfly species out of 17 are protected by the Habitats Directive, all butterfly species are Red-listed, and one third of the mammal species occur only on Red Lists. It can be concluded that indices for plants, mammals and butterflies are sensitive to the Red List criterion.

For mammals and butterflies, applying only the Red List criterion leads to different rankings, compared to other options. For mammals this can be explained by the fact that one third of the species in this group (i.e. river specific bats) do not occur on Red Lists, but are considered important by international and national hard law. Therefore, the omission of these species has a large influence on the model outcomes. Thus, indices for mammals are sensitive to exclusion of international valuation criteria. For butterflies, our results can not be explained by pointing out that many species are left out of the assessment when only the Red List criterion is applied, because all species are red-listed. The observed differences between valuation based on the Red List criterion only and all other options are probably due to the relatively small differences between the scores of the reconstruction alternatives. A small variation in reconstruction alternative scores already caused a different ranking.

The decision to exclude certain valuation criteria can lead to very different assessment outcomes. This is the case with sensitive groups. For example, in impact assessment for higher plants and butterflies, the decision to exclude the Red List criterion (i.e. to exclude endangered species) is an important decision, because of the large influence on the outcome of the assessment. For these two groups the Red Lists remain a highly important criterion for assessing impacts.

4.5 Further research

The model can easily be adapted in such a way that it can use data on species abundance as well. If sufficient data for the entire set of protected and endangered set of species becomes available, the model BIO-SAFE can be improved by incorporating additional environmental variables (e.g. moisture regime, acidity and nutrient levels), species specific minimum required ecotope surface area thresholds (e.g. minimum area requirements for a viable population) and spatial configuration of ecotopes. In addition, a more thorough validity test is required, using larger datasets that concern longer time periods and allow testing for each species group separately. This can make clear for which species groups additional variables must be implemented in order to make prediction of effects of habitat changes on species more accurate (Mac Nelly ***et al.***, 2003).

Further research must point out how much uncertainty can be attributed to specificity of species to ecotopes and value distribution within species groups. The main uncertainties of a biodiversity impact assessment are uncertainty in the data that are used, in the methodologies that are applied and in the value judgements provided by the experts. In this paper we focussed on the second and third source of uncertainty (Geneletti ***et al.***, 2003). Future research might also include uncertainty in data concerning ecotope distribution for different reconstruction alternatives and in data on species presence. The selection of riverine species for the model BIO-SAFE is partly based on expert judgement. A more rigorous method for species selection would be based on large data sets and mathematical or logical algorithms. However, this data is insufficiently available (De Nooij ***et al.***, 2004, Buijse ***et al.***, 2005). The data required would cover species autecology, biogeography and life strategies of species from many different taxonomic groups, and data on patterns and processes of river floodplain ecosystems.

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Chapter V

Protected and endangered species and hydrodynamics

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Abstract

This paper examines the relationship between protected and endangered riverine species (target species) and hydrodynamics in river-floodplain ecosystems, combining ecological and policy-legal aspects of biodiversity in river management. The importance of different hydrodynamic conditions along a lateral gradient was quantified for various taxonomic groups.

Our results show that (i) target species require ecotopes along the entire hydrodynamic gradient; (ii) different parts of the hydrodynamic gradient are important to different species, belonging to different taxonomic groups; (iii) in particular low-dynamic parts are important for many species and (iv) species differ in their specificity for hydrodynamic conditions. Many species of higher plants, fish and butterflies have a narrow range for hydrodynamics and many species of birds and mammals use ecotopes along the entire gradient.

Even when focussing only on target species, the entire natural hydrodynamic gradient is important. This means that the riverine species assemblage as a whole can benefit from measures focussing on target species only. River reconstruction and management should aim at re-establishing the entire hydrodynamic gradient, increasing the spatial heterogeneity of hydrodynamic conditions.

1 Introduction

Natural river-floodplain ecosystems exhibit a hydrodynamic gradient from the main channel to inundation-free areas. A wide variety of riverine habitats exists along this gradient, in space and time, created by the dynamic interaction of water, sediment and biota, leading to high biodiversity (Bayley, 1995; Ward ***et al.***, 2002). Species characteristic of the river-floodplain ecosystem (hereafter termed riverine species) have adapted their life histories to match riverine conditions. However, many riverine species have become rare and endangered in the Rhine and Meuse catchments, as a consequence of the dramatic changes in river-floodplain ecosystems (e.g. the construction of dikes, dams, groynes and weirs, conversion of floodplains to agricultural land, water pollution and invasive species). These modifications have greatly reduced the spatial heterogeneity, as well as the variation in hydrodynamic conditions along the lateral gradient in the floodplain, but also along the longitudinal and vertical dimensions (Ward & Tockner, 1999; Aarts ***et al.***, 2004). Ecological rehabilitation aims at restoring riverine biodiversity by rehabilitating hydrodynamic and morphodynamic processes in river-floodplain ecosystems, and introducing semi-natural grazing regimes (Nienhuis ***et al.***, 2002). Ecological rehabilitation also includes improvement of water chemistry and the remediation of toxic river sediments (Leuven ***et al.***, 2005).

Large-scale reconstruction measures are being prepared and implemented in river basins of north-western Europe for the purpose of flood defence, ecological rehabilitation and infrastructural improvements (Van Stokkom ***et al.***, 2005). These measures will have far-reaching consequences for the physical structure and dynamics, and hence for the ecological functioning, of river-floodplain ecosystems (Nienhuis ***et al.***, 1998). Political and legal goals state the importance of ecological rehabilitation and provide regulations and time horizons. According to the European Water Framework Directive (Council Directive 2000/60/EC), for natural rivers a good ecological status, and for heavily modified waters a good ecological potential, have to be achieved by 2015.

The legislative framework for nature protection in Europe consists of the Habitats Directive and the Birds Directive. Significant negative impacts of human activities on species and habitats protected by these directives are not allowed, unless (i) there are no alternative solutions and (ii) there are imperative reasons of overriding public interest that demand these activities (Council Directive 79/409/EEC; Council Directive 92/43/EEC). Even if these two conditions have been met, the negative impacts on protected habitats have to be compensated for. River managers are therefore obliged to take protected species into account in their effect assessments for spatial planning, physical reconstruction and management (e.g. Environmental Impact Assessments and Strategic Environmental Assessments). Another important and widely used instrument in species conservation is that of Red Lists. In this paper, we use the term target species to refer to both legally protected and red-listed species.

Attuning the aims of flood defence, ecological rehabilitation and nature protection requires tools that integrate policy and legislation goals with ecological knowledge about target species in river-floodplain ecosystems. Expressing both actual and potential biodiversity values offers opportunities to assess the impacts of physical reconstruction on biodiversity. The BIO-SAFE model (Spreadsheet Application For Evaluation of BIOdiversity) is such a tool (De Nooij ***et al.***, 2004, 2005; Lenders ***et al.***, 2001).

The theoretical relation between hydrodynamics and biodiversity is well-known and is exemplified in the flood pulse concept (e.g. Junk ***et al.***, 1989). Empirical evidence supports this concept and shows that different taxonomic groups utilize the gradient differently (Van den Brink, 1994; Van den Brink ***et al.***, 1996; Ward & Tockner, 2001; Chovanec ***et al.***, 2005). However, to what extent the relation between hydrodynamics and biodiversity is also valid for target species is largely unknown. Maximizing ecological benefits of floodplain reconstruction, and minimizing conflicts between river management and nature protection, require knowledge on the response of protected and red-listed riverine species to river dynamics. The BIO-SAFE model integrates available knowledge about habitat demands of these target species for the rivers Rhine and Meuse with their political and legal status. This model was used to answer the following questions:

1. How important are different parts of the hydrodynamic gradient for target species of different taxonomic groups in river-floodplain ecosystems?
2. How specifically do target species utilize the various parts of the hydrodynamic gradient?
3. What are the implications of the response of riverine target species to hydrodynamic conditions for river management?

2 Materials and methods

2.1 Model description

BIO-SAFE is a valuation model which links ecotopes to riverine target species listed in the European Habitats Directive, the European Birds Directive, the Conventions of Bern and Bonn and Red Lists. Ecotopes are defined as spatial units of a certain extent, which are relatively homogeneous in terms of vegetation structure, succession stage and the main abiotic site factors that are relevant to plant growth (Klijn & Udo de Haes, 1994). BIO-SAFE describes the habitat of riverine target species in terms of riverine ecotopes, derived from the Water Ecotope Classification published by Van der Molen ***et al.*** (2003), which includes the River Ecotope System (RES) by Rademakers & Wolfert (1994). The classification by Van der Molen ***et al.*** (2003) for rivers is based on vegetation structure and composition, inundation frequency (hydrodynamics), morphodynamics and land use. In BIO-SAFE, ecotopes are distinguished at four levels of scale (1:100,000; 1:50,000; 1:25,000; 1:10,000). At the finest level of scale (1:10,000), 60 different ecotopes are distinguished. River engineers, landscape ecologists and landscape designers use ecotopes in hydraulic models, landscape ecology and landscape design, making the concept of ecotopes a suitable tool for communication between the various disciplines active in river management. The model incorporates both natural ecotopes and man-made ecotopes.

Species were selected based on their occurrence in river-floodplain ecosystems. This included species characteristic of the current situation, but also of natural river-floodplain ecosystems. Selected species were subsequently attributed to the different ecotopes. The information on species and habitats was derived from a thorough literature survey, supplemented by expert knowledge. Taxonomic groups included in the model are higher plants, birds, herpetofauna (amphibians & reptiles), mammals, fish, butterflies and odonates (dragonflies & damselflies). In linking species to ecotopes the habitat demands of all life cycle stages were considered. Species in other taxonomic groups were either not listed as target species or were not characteristic of the Rhine and Meuse river-floodplain ecosystems. Table 1 lists the numbers of species included in each taxonomic group.

To each species, values were assigned on the basis of its policy and legislation status. Through the linkage of species to ecotopes, values were assigned to ecotopes as well (Figure 1). An explanation of the species selection process, the value assignment, the development of the ecotope typology and the linkage of species to ecotopes, including a full description of the functionalities of the BIO-SAFE model, can be found in Lenders ***et al.*** (2001) and De Nooij ***et al.*** (2001, 2004).

Table 1. Numbers of target species and ecotopes (between brackets) included in BIO-SAFE for each taxonomic group and per hydrodynamic class. Criteria for the classification of hydroclasses were modified from Van der Molen ***et al.*** (2003). A full list of the species included can be found in De Nooij ***et al.*** (2001)

Hydro-class	Criterion	HP (n=136)	BI (n=60)	HF (n=9)	MA (n=9)	FI (n=20)	BU (n=17)	OD (n=6)	Total (n=257)
1	Deep water (>1.5 m)	0 (0)	22 (2)	3 (1)	3 (2)	14 (2)	0 (0)	0 (0)	42 (2)
2	Permanently flooded (<1.5 m)	4 (6)	31 (9)	3 (7)	5 (9)	18 (9)	3 (1)	4 (5)	68 (10)
3	River bank	18 (7)	36 (10)	3 (5)	8 (9)	14 (10)	3 (1)	4 (6)	86 (11)
4	Flooded >100 $d.yr^{-1}$	25 (12)	56 (13)	9 (12)	8 (13)	8 (4)	6 (6)	6 (10)	118 (13)
5	Flooded 20-100 $d.yr^{-1}$	105 (22)	53 (22)	9 (20)	8 (20)	6 (3)	17 (13)	5 (14)	203 (23)
6	Flooded <20 $d.yr^{-1}$	112 (23)	53 (23)	9 (22)	8 (22)	6 (6)	15 (12)	5 (14)	208 (25)
7	Never flooded*	54 (13)	46 (12)	9 (11)	7 (12)	6 (5)	10 (5)	5 (6)	137 (14)

* Note that in the absence of flooding, aquatic ecotopes can still be present.

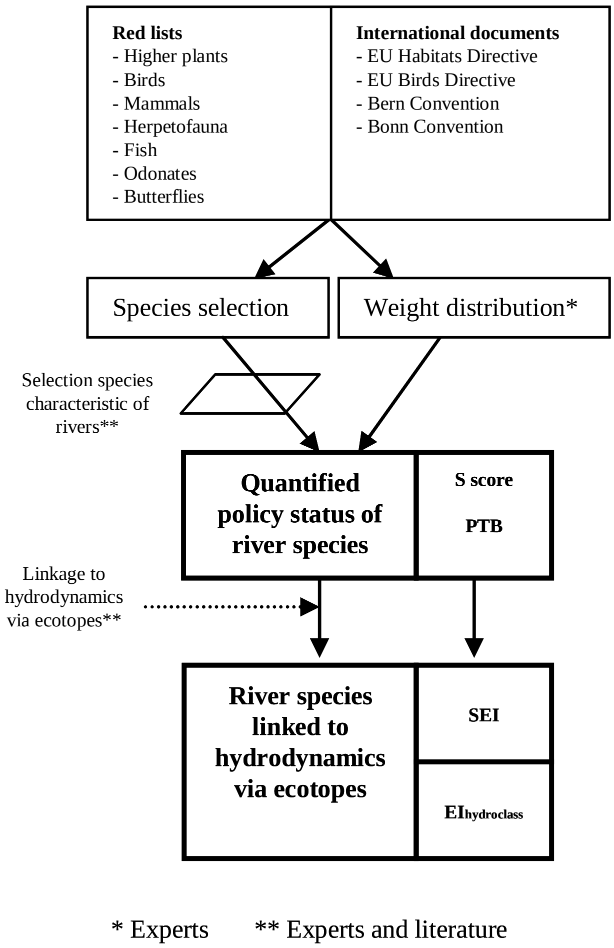

Figure 1. Schematic overview of the construction of BIO-SAFE and quantification of the potential of hydrodynamic conditions for target species. S score: Species-specific score, quantifying policy relevance; PTB: Potential Taxonomic group Biodiversity constant; SEI: Species-specific Ecotope Importance; relative contribution of species to the maximum potential value for that ecotope; THI: Taxonomic group Hydroclass Importance; potential value of the hydrodynamic class.

2.2 Data analysis and calculations

In this study, species selection, linkage to ecotopes and value assignment concern lowland river-floodplain ecosystems in the Netherlands and the Dutch political-legislative context. All analyses were carried out at the finest level of scale, as preliminary analyses had indicated that ecotopes defined at coarser levels were less accurate in describing the habitats of many species. The ecotopes were classified into seven different hydrodynamic classes along the hydrodynamic gradient (hydroclasses, Table 1). Note that an ecotope can occur in more than one hydroclass, such as the ecotope called natural levee pasture, which can occur under flooding conditions with a total inundation time of less than 100 days per year (i.e. hydroclasses 5 and 6; Table 1).

Based on the values assigned to the species, the linkage of the species to the ecotopes and the relation between hydrodynamic conditions and the occurrence of ecotopes (Table 1), we quantified the biodiversity potential of a hydrodynamic class (Hydroclass Importance) (Figure 1). This procedure was applied to all taxonomic groups combined (Figure 2) and to each taxonomic group separately (Figure 3). For all taxonomic groups combined, the total score in all hydroclasses was set at 100% for each taxonomic group (in order to weight the different taxonomic groups equally, i.e. irrespective of the total number of species in a taxonomic group). In order to calculate the importance for each ecotope, the species scores were first summed per taxonomic group, resulting in the ***Potential Taxonomic group Biodiversity constant*** (PTB):

$$\text{PTB} = \sum \text{species scores (for all species per taxonomic group)} \qquad [1]$$

Subsequently, the score of a species was assigned to the ecotopes it was linked to (species score). This species score was divided by the PTB, resulting in the ***Species specific Ecotope Importance*** ($\text{SEI}_{\text{ecotope i}}$) which is the species' relative contribution to the maximum potential value of that ecotope.

$$\text{SEI}_{\text{ecotope i}} = \text{species score / PTB} \qquad [2]$$

For each taxonomic group, the SEI values of all species were summed per hydroclass (Table 1) in order to calculate the ***Taxonomic group Hydroclass Importance*** (THI). This is defined as the potential value of that hydrodynamic class as a habitat for riverine target species belonging to a certain taxonomic group.

$$\text{THI}_{\text{hydroclass k}} = \sum \text{SEI (for all species per taxonomic group occurring in ecotopes with hydrodynamic class k)} \qquad [3]$$

Per hydrodynamic class, ecotopes were only used in the calculation if (i) they occurred under those hydrodynamic conditions and (ii) they were used by at least one species in that taxonomic group (see Table 1), in order to exclude ecotopes which are unsuitable for that taxonomic group (e.g. fish do not occur in terrestrial ecotopes).

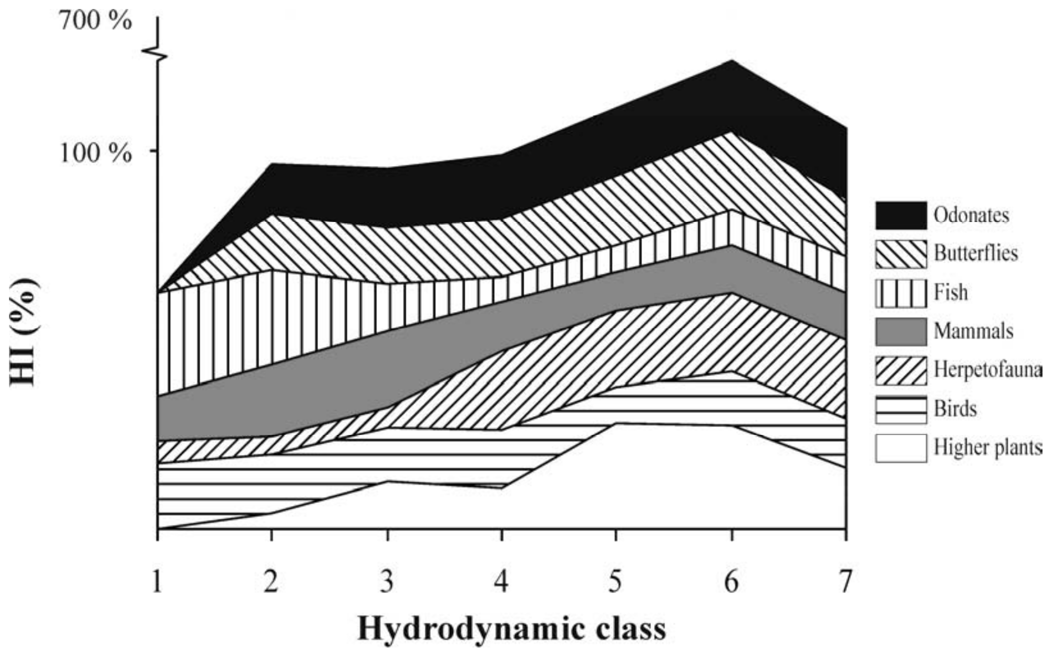

Figure 2. Hydroclass Importance (HI), for all species groups combined, along the hydrodynamic gradient. The contribution of each taxonomic group was set at 100%. 1: highest degree of hydrodynamics, 7: lowest degree of hydrodynamics.

3 Results

When all taxonomic groups of target species are combined, the biodiversity potential shows an increase with decreasing hydrodynamics, until class 6, where an optimum is reached (Figure 2). This class also harbours the largest number of target species (Table 1). The contribution of the different taxonomic groups to the potential differs markedly along the hydrodynamic gradient. The results per taxonomic group are given in Figure 3.

For higher plants, the low-dynamic parts (classes 5 and 6) are most important, although a number of plants are confined to class 3, viz., strapwort (***Corrigiola litoralis***), tall pepperwort (***Lepidium graminifolium***) and bur medick (***Medicago minima***). Many plant species have a narrow range (high specificity) along the hydrodynamic gradient.

Fish show a pattern opposite to that of plants, in which the high-dynamic parts (classes 1 and 2) are most important. Sturgeon (***Acipenser sturio***), twaite shad (***Alosa fallax***) and lampern (***Lampetra fluviatilis***) are restricted to the high-dynamic parts. Species with a broad range are predominantly found in the low-dynamic parts (classes 5 – 7); these include weatherfish (***Misgurnus fossilis***), eel (***Anguilla anguilla***) and crucian carp (***Carassius carassius***).

For birds, the entire hydrodynamic gradient is important. Species with a narrow range (high specificity), such as black stork (***Ciconia nigra***), curlew (***Numenius arquata***) and redshank (***Tringa totanus***) are mainly found in the low-dynamic parts (classes 4– 7). Many bird species have a broad range for the hydrodynamic gradient; these include waterfowl like several duck species, black tern (***Chlidonias niger***) and common tern (***Sterna hirundo***).

The extremely dynamic parts (class 1) are of no importance to butterflies, while the high-dynamic parts (classes 2 and 3) are important to brown argus (***Aricia agestis***), queen of spain frittillary (***Issoria lathonia***) and glanville frittillary (***Melitaea cinxia***). Species restricted to the low-dynamic parts (4-7) include scarce large blue (***Maculinea teleius***), chequered skipper (***Carterocephalus palaemon***) and silver-washed frittillary (***Argynnis paphia***).

For herpetofauna, it is the low-dynamic parts (classes 4 – 7) that are the most important by far. The high-dynamic parts (1 – 3) are only used by grass snake (***Natrix natrix***), lake frog (***Rana ridibunda***) and edible frog (***Rana kl. esculenta***). These species occur across the entire hydrodynamic gradient.

Odonates, like butterflies, are absent from the extremely dynamic parts. Gomphidae use almost the entire gradient, while green hawker (***Aeshna viridis***) and hairy dragonfly (***Brachytron pratense***) are restricted to the low-dynamic parts (classes 4 – 7).

For mammals, the entire hydrodynamic gradient is important, although compared to birds, the high-dynamic parts (classes 2 and 3) are more important to them. Species with a narrow range for the hydrodynamic gradient include red deer (***Cervus elaphus***), root vole (***Microtus oeconomus***) and water shrew (***Neomys fodiens***). Species with a broad range for the hydrodynamic gradient include beaver (***Castor fiber***), otter (***Lutra lutra***) and pond bat (***Myotis dasycneme***).

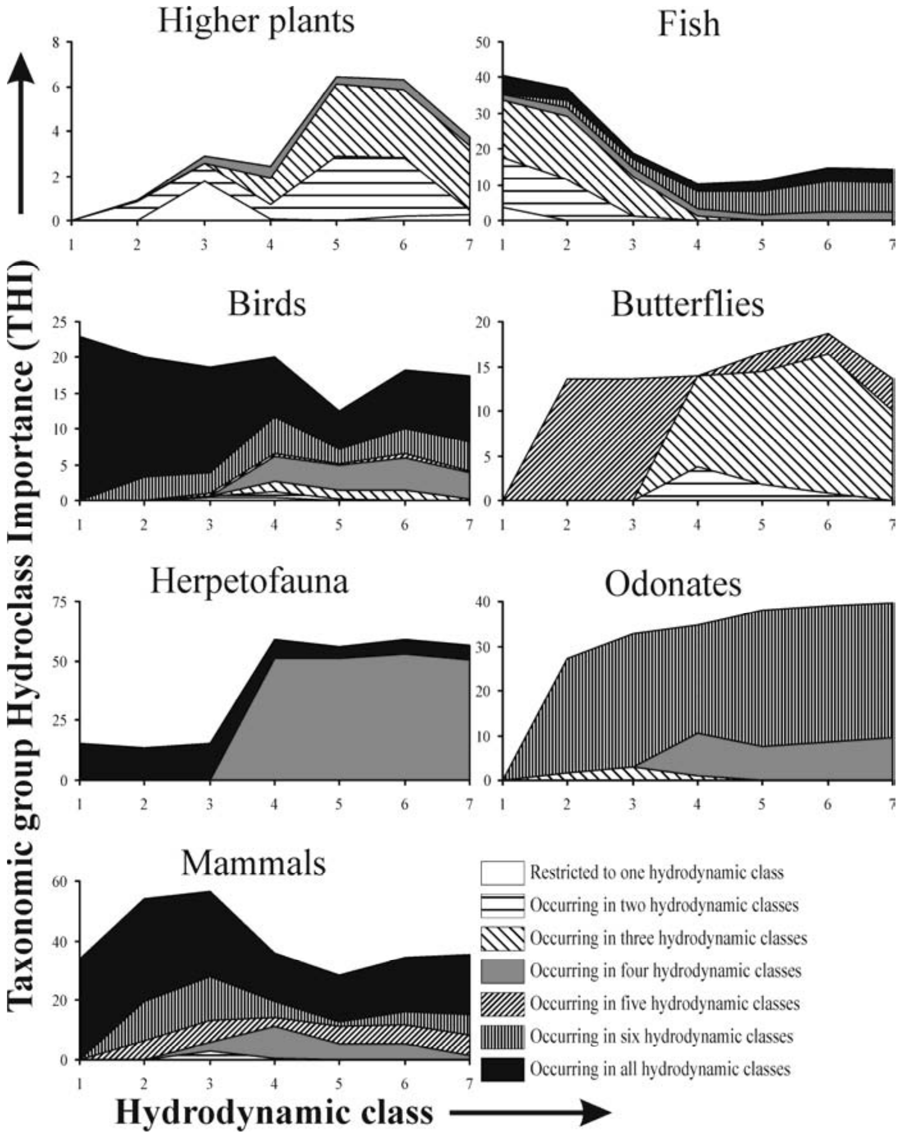

Figure 3. Taxonomic group Hydroclass Importance (THI) for the different species groups along the hydrodynamic gradient. Species were grouped according to their specificity for hydrodynamic conditions, i.e., the number of different hydrodynamic classes in which they can be found. 1: highest degree of hydrodynamics, 7: lowest degree of hydrodynamics.

4 Discussion

Traditionally, river management has focussed mainly on the chemical and physical aspects of river-floodplain ecosystems, such as water quality, water quantity and flow power. Safety and economic development (in terms of flood defence, food and drinking water quality, infrastructure, agriculture, mineral extraction, etc.) were always the main goals. River managers today are faced with highly modified river-floodplain ecosystems, with large numbers of species that are nowadays protected and/or endangered and therefore require special attention. The challenge in the near future is to reconstruct and manage these river-floodplain ecosystems in a way that reconciles flood risk management, infrastructural works and economic development with ecological rehabilitation (Nienhuis & Leuven, 2001), within legal boundaries imposed by nature conservation legislation. River-floodplain ecosystems are structured by hydrodynamics, morphodynamics and vegetation succession. River managers should be able to influence these processes in such a way as to achieve the various goals of river management. This requires knowledge about the causal relationships between physical processes and desired endpoints. In the present study, target species (endpoints of nature conservation) were linked to hydrodynamics. Our results therefore integrate the ecological significance of hydrodynamic conditions with the relative importance of riverine species in policy and legislation.

Although target species occur along the entire gradient, the potential for these species increases gradually with decreasing dynamics and reaches an optimum when the inundation frequency is between 0 and 20 days per year (class 6). In addition, each taxonomic group shows a different distribution along the gradient and responds to hydrodynamics in a typical and different fashion. For some groups (e.g., rheophilous fish), it is the high-dynamic parts that are more important, while for other groups (e.g. higher plants, herpetofauna, butterflies), the most important parts are those with low dynamics. Many species of birds, mammals and odonates predominantly use the entire gradient, while most plant species and – to a lesser extent – butterflies and fish are specifically bound to one or two hydrodynamic classes.

Species found along the entire gradient may be indifferent to hydrodynamics, but may also specifically utilize different parts of the gradient for different activities (e.g. foraging, breeding and resting) and therefore depend on the entire gradient in order to successfully complete their lifecycle (Verberk & Esselink 2003). Usually, the latter situation is the case. For example, black terns (***Chlidonias niger***) are very specific as regards their breeding site, using floating rafts of terrestrializing vegetation, such as those formed by water soldier (***Stratiotes aloides***), but forage in a broad range of ecotopes (from the main channel and lakes to marshland vegetation), feeding mainly on small fish and large aquatic insects, such as dragonflies. The grass snake (***Natrix natrix***) is also found along the entire hydrodynamic gradient, but selectively requires low-dynamic ecotopes with conditions making them suitable as hatcheries for their eggs, while many other ecotopes are suitable foraging or resting sites. The water bat (***Myotis daubentonii***) selectively catches its prey above open water, and is specific with regard to its resting and breeding habitat (e.g. old growth trees). The river darter (***Gomphus flavipes***) specifically uses submerged sandy river banks during its larval stages, whereas upon emerging, the adult uses a variety of terrestrial ecotopes for maturation, roosting and foraging.

These examples illustrate the limitations of the concept of ecotopes in describing habitats when applied to mobile animals. This was to be expected, as the ecotope classification concept we used was primarily based on factors structuring plant communities. Although many important causal factors for animal species are not incorporated in the ecotope classification applied in this study, this concept can to a certain extent be applied to quantify biodiversity potential (Lenders ***et al.***, 1998).

Our study dealt only with riverine target species of various taxonomic groups of the Rhine and Meuse in the Netherlands. These river-floodplain ecosystems are highly managed lowland rivers, potentially limiting the extrapolation of the results to other systems. However, species and ecotopes incorporated in the model were derived from both natural and modified river-floodplain ecosystems (see Materials and Methods). Moreover, the results are largely in accordance with those of studies dealing with various aquatic taxonomic groups of river-floodplain ecosystems along the lower Rhine and Meuse (Van den Brink, 1994; Van den Brink ***et al.***, 1996), with total species richness in the Danube (Ward & Tockner, 2001), with a weighted biodiversity score in the Danube (Chovanec ***et al.***, 2005) and with fish biodiversity in general (Aarts ***et al.***, 2004). This is a strong indication that, for river-floodplain ecosystems, (i) our results may have generic meaning and (ii) the riverine species assemblage as a whole can benefit from measures focussing on creating suitable conditions for target species only. These results are important in the light of the implementation of many measures as required by the EU's Water Framework Directive (WFD).

Our results show that for target species the entire hydrodynamic gradient found in a natural riverine landscape is important. Our results also show that both aquatic and terrestrial ecotopes in the low-dynamic parts of the hydrodynamic gradient are particularly valuable. The importance of low-dynamic aquatic parts was also highlighted by a study by Van den Brink ***et al.*** (1996).

Measures aimed at flood defence, which include lowering of floodplains and river dike diversion (winter bed enlargement), may provide opportunities as well as threats for (protected) biodiversity. The habitat demands of riverine species in relation to hydrodynamics should set the boundary conditions for physical reconstruction and management aimed at combining safety goals with ecological rehabilitation and nature protection (Nienhuis & Leuven, 2001; Van Stokkom ***et al.***, 2005). Increasing the opportunities for target species requires enlargement of the winter bed (i.e., more space for the gradient to develop), a prerequisite running counter to current reconstruction plans. Because space is scarce, river managers are looking for room for water discharge in the vertical dimension, by riverbed deepening and floodplain lowering (Nienhuis & Leuven, 1998). This causes higher levels of hydrodynamics between the dikes, and thus does not result in restoration of the hydrodynamic gradient, but in a loss of low-dynamic parts (and the creation of a 'bathtub' situation). Our results indicate that these parts are of vital importance, so that situations with limited space require tailor-made designs. These designs need to combine our results on the importance of different hydrodynamic conditions for riverine species with more specific knowledge about their demands in terms of size and configuration of habitat elements (e.g., ecotopes; Wiens, 2002). For example, extra space for low-dynamic ecotopes can be created by over-dimensioning of flood defence measures such as lowering of floodplain and widening or digging of secondary channels. Tailor-made designs also require a sound inventory and assessment of the actual situation and potentials specific for that location.

In conclusion, even when focussing only on target species, no part of the natural hydrodynamic gradient can be neglected in reconstruction and restoration designs. River reconstruction and management should aim at enlarging the winter bed in order to re-establish the hydrodynamic gradient. When this is not possible, spatial and temporal heterogeneity of hydrodynamic conditions should be maximised within the spatial limits.

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Chapter VI

Relating the ecological and legal framework

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Abstract

Ecology and legislation are two essential frameworks for nature conservation. Ecological knowledge has been implemented in legislation to protect species and their habitats, and the two frameworks meet in the practice of nature conservation in legal procedures that require ecological information. However, differences in approaches towards nature, and terminology, cause several problems. Moreover, from an ecological-scientific perspective nature conservation legislation and jurisprudence often seems illogic and incomplete. Conversely, ecological science often can not provide the answers to legal questions and the certainty that is required in legal procedures. This article analyses the relation between ecological knowledge and legal instruments for nature conservation. Special attention is paid to possibilities and limitations of the legal framework for nature conservation in river management.

It is concluded that ecological reality is much more complex than the legislator has implemented in the current legal framework and that several recent ecological insights have not yet or insufficiently been implemented (such as the importance of ecosystem dynamics, heterogeneity, non-linear behaviour and uncertainty). This causes both enormous information needs and many too restricted criteria for determining effects on species and ecosystems. The selection of protected species and legal procedures for their protection frequently are too limited from an ecological point of view. Definitions of many ecological terms in legislation are sometimes vague and they often deviate from generally accepted ones in ecological sciences.

In order to make legislation more appropriate for nature conservation and ecological research more relevant for legal procedures, more attention must be paid to the ecological relevance and extinction risk of species and ecosystems. In management plans, codes of conduct, and jurisprudence, more attention must be paid to ecosystem dynamics and key processes, ecological scale and context, spatial heterogeneity and coherence of ecological networks. Furthermore, jurisprudence should anticipate more appropriate on uncertainties in effect assessments. Effect assessments should consider cumulative effects and must take place at the most relevant level, which is usually the regional metapopulation network. Within the ecological field, more insight is required concerning the distribution of protected species, their habitat (the relation between organism and environment), the other species they depend on, and the response of species and ecosystems to human activities.

The jurists and decision makers who apply the current nature conservation legislation should try and understand more of an ecological approach to nature conservation. Ecologists should realise that legislation represents a world of thought in its own right, with its own objectives and criteria for making valid claims.

1 Introduction

1.1 Problem definition

Many species and habitats in the Netherlands have become rare or have entirely disappeared (Van Nieukerken & Van Loon, 1995). The main causes are anthropogenic habitat destruction and fragmentation, deterioration of soil, water and air quality, hunting, and exploitation of species (Heywood & Watson, 1995).

Legislation provides the necessary instruments for administrative-legal protection of nature. Ecology is an important science for providing the required knowledge and information on effects of human activities on species and their habitats. Because of the increased influence of legal instruments for nature conservation on the decision-making concerning nature and landscape, the need for the input of ecological knowledge and information increases (Verschuuren & Van Wijmen, 2003). Insight in the relation between approaches and procedures of legislation on the one hand, and ecology on the other, is very important for optimal functioning of nature conservation (Backes, 2004).

In practice, however, compliance of human activities with the obligations of nature conservation legislation causes major problems in the Netherlands. Many projects aimed at the realisation of new infrastructure, housing, industrial development, and nature development have been hampered by unclarities about legal obligations as well as ecological effects (Backes, 2004; Neumann & Woldendorp, 2002). Furthermore, it is often stated that legal regulations for species protection are not appropriate (Broekmeyer ***et al.***, 2003): (1) because of the emphasis on individuals many legal procedures are ecologically speaking unnecessarily carried out, and (2) because of ecologically limited prohibition provisions many plans with possibly negative impact on protected species and areas do not enter legal procedures.

The legislation is so wide of set-up that ecology cannot satisfy all information needs (Vos ***et al.***, 2002). In addition, ecologists state that the way in which species and ecosystems are approached in the nature conservation legislation is too limited and therefore inappropriate (Capelle & Stumpel, 2003; Broekmeyer ***et al.***, 2003). Ecologists are having difficulties to answer the legal key questions, whereas jurists, economists and spatial planners regularly have insufficient attention for the complexity of ecological issues.

The abovementioned problems arise from legal obstacles, complexity of the legislation and lack of attention to or familiarity of decision makers with ecological and legal aspects (Bastmeijer & Verschuuren, 2004). Also the attention paid to, and familiarity with legislation among ecologists is often limited. Other causes relate to ecological knowledge and information: the lack of actual information on the distribution, the population size and dynamics of protected flora and fauna, as well as a lack of insight into the response of protected species and ecosystems to human activities. The problems are made worse by the vagueness of quality standards for ecological research and the lack of an unambiguous methodology for ecological impact assessments.

Another very important source of problems is the relation between ecology and legislation. Many important terms relating directly to ecological reality (e.g. significant effect, favourable conservation status) are legally ambiguous and not easy to handle in practice (Capelle & Stumpel, 2003; Broekmeyer ***et al.***, 2003). There are conflicts between the legal and ecological conceptual frameworks, owing to differences in aims, approaches and terminology. These differences must be made clear in order to provide a good basis for optimal nature conservation.

1.2 Goals and research questions

The goal of this paper is to describe the implementation of ecological knowledge in legal instruments for nature conservation, to trace problems in the relation between ecological and legal approaches, and to recommend options for improvement of legal instruments. It is also intended to provide both jurists and ecologists with helpful insights into the relation between ecology and legislation. The following questions are raised:

1. How is ecological knowledge of species and ecosystems implemented in legal instruments for nature conservation?
2. What aspects of legislation are too wide, and what aspects of legislation are too narrow, according to ecological criteria?
3. What are the consequences of the differences between ecology and legislation?
4. What are opportunities for improvement of the relation between ecology and legislation?

The ecological and the legal framework for nature conservation both have their own criteria for making valid claims. In this paper we criticise legislation from an ecological point of view. This criticism can, however, concern only the ecological dimension of legal instruments. In order to function, legislation and jurisprudence have to account for criteria such as legal security, providing clear frameworks to maintain law and order, and internal consistency of the law. We are aware that, in many cases, these criteria may limit full implementation of the available ecological body of knowledge.

1.3 Methodology and outline

This paper firstly analyses contents and procedures of the legal framework, focussed on approaches towards nature and species selection, procedures, and term definitions (paragraph 2). This analysis is based on documents that contain legislation and legal procedures relevant for the Netherlands. The framework for nature conservation legislation within the European Union, formed by the Birds Directive (BD; Council Directive 79/409/EEC) and the Habitats Directive (HD; Council Directive 92/43/EEC), contains obligations for the Member States concerning both species protection and area protection. Species protection has been implemented into Dutch law by means of the Flora and Fauna Act (FFA); area protection in the Dutch Nature Protection Act 1998 (NPA). During this research the area component of the European legislation was not yet fully implemented in the Dutch NPA. Therefore, the present analysis of area protection is only based on the HD and BD.

In paragraph 3, the ecological framework is also analysed regarding its approaches towards nature and species selection, its procedures and its term definitions. This is based on ecological textbooks and other scientific publications, and interviews with ecological scientists (De Nooij ***et al.***, 2006a). Paragraph 4 compares the ecological and legal frameworks and focuses on aspects of legislation which are too limited or too far-reaching according to ecological criteria. The consequences of these differences are briefly highlighted. This paragraph also analyses jurisprudence, because in legal procedures ecological terms are interpreted and additional ecological aspects are included. In total, 27 verdicts of the Court of Justice of the European Communities (CoJEC), the Dutch Administrative Jurisdiction Department of the Council of State (ABRvS; Afdeling Bestuursrechtspraak Raad van State) and Dutch Courts of Justice (Rb; Rechtbank) were evaluated regarding frequency and interpretation of ecological terms.

The consequences of differences between ecology and legislation are studied in more detail within the context of river management. River management provides interesting case material because of the upcoming large scale physical reconstruction of river floodplains for hydraulic infrastructure, flood risk management and ecological rehabilitation (Nienhuis ***et al.***, 2002; Leuven ***et al.***, 2002; Van Stokkom ***et al.***, 2005). Problems with nature conservation legislation can be anticipated because of expected effects on protected species and areas in complex dynamic systems. The effects will be massive and can be positive as well as negative. Impact assessment will be complicated and may include major uncertainties. Three verdicts in legal procedures relating to river floodplains are analysed in more detail.

Paragraph 5 summarises the relation between the ecological and legal framework, and gives opportunities for improvement.

2 Legal framework

2.1 Objectives and approaches to nature and species selection

Species protection FFA

The FFA aims at a general level of species protection. The ultimate goal is conservation and restoration of the variability of wildlife, recognising species as functional elements of ecosystems. This is also defined as the wish to conserve the genetic variability and species richness. Moreover, the opening words of the FFA explicitly mention the intrinsic value of animals as a reason for protection.

The FFA emphasises passive protection of species by means of a general prohibition scheme. Special attention is paid to threatened species and species for which the Netherlands has international responsibility (DHRSG, 1996). Article 1 of the FFA concerns all ontogenetic stages of wild animals and plants.

According to the FFA all vertebrates are protected (fish, amphibians, reptiles, birds and mammals), except a number of species mentioned in a negative list (e.g. Black rat, Brown rat, House mouse and as far as they are domesticated, the European polecat, the European rabbit and the Pig) and fish species which are dealt with in the Fisheries Act. Furthermore, protected species are designated in a positive list. This applies to plants (higher plants only) and a limited number of invertebrates (e.g. 26 of in total 81 native butterfly species and 4 species of approximately 4300 ants, bees and wasps). This leaves all mosses, lichens and mushrooms, and most invertebrates unprotected, irrespective of their extinction risk or ecological function. Provisions applying to protected species mainly concern prohibition of removing plants from their habitat and killing, catching, owning and trading of animals, and to deliberate disturbance of animals and/or their habitat (articles 8-12).

Important is that the FFA prohibition provisions are aimed at the individual animal or the individual plant. As mentioned above, for animals the recognition of intrinsic value is also a reason for protective measures. This means that, for animals, sustainability of populations is not the only aim of protection. Protection of individual flowers and seeds is considered important because it is difficult to determine which quantity of flowers and seeds can be harvested without objection (DHRSG, 1993).

Human activities rapidly result in offence of the prohibition provisions, whereupon dispensation is requested. Dispensation is only possible if the ‘favourable conservation status’ of the species is not jeopardised. The favourable conservation status is related to the population level (box 1). Although the prohibition provisions for protected species primarily focus on individuals, judgements of acceptability of human activities are based on population effects.

Box 1: Definition of key terms of the Habitats Directive (article 1).

Habitat of a species
An environment defined by specific abiotic and biotic factors, in which the species lives at any stage of its biological cycle.

Natural habitats
Terrestrial or aquatic areas distinguished by geographic, abiotic and biotic features, whether entirely natural or semi-natural.

Favourable conservation status
The conservation status of a species will be taken as "favourable" when:

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

The conservation status of a natural habitat will be taken as "favourable" when:

- Its natural range and areas it covers within that range are stable or increasing, and
- The specific structure and functions which are necessary for its long-term maintenance exist and are likely to continue to exist for the foreseeable future, and
- The conservation status of its typical species is favourable.

Area protection by the BD and HD

The aim of the BD and HD is: (1) to contribute towards ensuring bio-diversity through the conservation of natural habitats and of wild fauna and flora in the European territory of the Member States to which the Treaty applies and (2) to maintain or restore, at favourable conservation status, natural habitats and species of wild fauna and flora of Community interest.

Both directives emphasise ecological-scientific underpinning of population sizes, selection of species and areas, and evaluation of impacts of human activities on protected species. Moreover, article 2 of the BD also pays attention to cultural, economic and recreational demands regarding population size. Article 1 of the HD indicates that selection of protected areas (Special Areas of Conservation; SACs) requires special attention to the species that are:

1. Endangered, except those species whose natural range is marginal in that territory and which are not endangered or vulnerable in the western Palaearctic region; or
2. Vulnerable, i.e. believed likely to move into the endangered category in the near future if the causal factors continue operating; or
3. Rare, i.e. with small populations that are not at present endangered or vulnerable, but are at risk. The species are located within restricted geographical areas or are thinly scattered over a more extensive range; or
4. Endemic and requiring particular attention by reason of the specific nature of their habitat and/or the potential impact of their exploitation on their habitat and/or the potential impact of their exploitation on their conservation status.

Article 4.1 of the BD gives similar criteria. Article 3 of the HD explains the approach for area protection. A coherent European ecological network of SACs shall be set up under the title ‘Natura 2000’. This network, composed of sites hosting the natural habitat types listed in

Annex I and habitats of the species listed in Annex II, shall enable the natural habitat types and the species' habitats concerned to be maintained or, where appropriate, restored at a favourable conservation status in their natural range (box 1). The Natura 2000 network shall include the SACs classified by the Member States pursuant to the BD. Where they consider it necessary, Member States shall endeavour to improve the ecological coherence of Natura 2000 by maintaining, and where appropriate, developing features of the landscape, which are of major importance for wild fauna and flora.

Article 6 of the HD (box 2) indicates that conservation measures must be taken to conserve the quality and quantity of SACs. Furthermore, a procedure is given for assessing activities which can possibly affect SACs.

2.2 Procedures and terminology in the legal framework

Species protection FFA

In order to determine whether the FFA will have consequences for an activity, first of all a (field) survey concerning the presence of protected species by must be carried out in the area affected by the human activities. If protected species are present, it must be examined which prohibition provisions may be violated (articles 8 till 12). For example, article 8 forbids removal of protected plants from their growing sites. Articles 10 and 11 forbid the disturbance of individual animals and their nests, lairs, reproduction sites, or resting places, respectively. If one of these provisions may be offended, one must apply for dispensation on the basis of article 75, unless an exemption regulation applies.

Species protection legislation considers three different species categories with an increasing level of protection:

1. Frequently occurring protected species of the groups mammals, reptiles, amphibians, ants, snails and vascular plants (Red Data List category ‘least concern’; Red Data Lists are documents that classify species into different categories of threat based on data concerning rarity and trend in abundance and geographical distribution; IUCN, 1993;1994).
2. Protected species of the groups mammals, reptiles, amphibians, fish, butterflies, beetles, crustaceans and vascular plants, which are slightly threatened (Red Data Lists: ‘near threatened’).
3. Species of the groups mammals, reptiles, amphibians, fish, butterflies, dragon- and damselflies, molluscs and vascular plants listed in Annex IV of the HD and many species that classified in the Red Data Lists as ‘extinct in the wild in the Netherlands’, ‘critically endangered’, ‘endangered’ or ‘vulnerable’. Furthermore some species were added for societal reasons e.g. the badger. Category 3 also comprises all bird species.

The three species categories correspond only partly to the degrees of threat given by Red Data Lists. Moreover, the classification differs per taxonomic group. There are two kinds of exemptions of the obligation to apply for dispensation (Dutch Ministry of Agriculture, Nature and Food Quality, 2005):

1. A generic exemption for species of category 1, if certain criteria are met;
2. An exemption provided that one acts in accordance with an approved code of conduct, for the species of the categories 2 and 3.

If no exemption can be obtained, a relatively limited effect assessment must be carried out for species of categories 1 and 2. To be able to get dispensation for species of category 3, a more extensive assessment is required. Both assessment frameworks demand that no harm will be done to the favourable conservation status of a protected species. For species of category 3, it is required that alternatives for the activity do not exist and that the activity complies with a limitative list of interests.

The codes of conduct must explain why the particular activities do not have substantial effects on protected species. Therefore the effects of these activities on possibly affected populations must be examined. In case of population effects, the regional, national and European population must be considered. In order to determine the appropriate levels of scale, three types of populations are distinguished (DHRSG, 2004a):

1. Isolated populations, for which the local level must be considered;
2. Subpopulations of metapopulations, for which also higher levels of scale, i.e. other subpopulations that are connected to the local population, must be taken into account;
3. Metapopulations, in case the local population already is the entire global population.

It can be concluded that an activity has no substantial effects if resistance or resilience of (the population of) a species on the short or long term is sufficient for maintaining the favourable conservation status (box 1). Although not explicitly mentioned in the FFA and related Ministerial Decrees, the responsible minister stated that effect assessment must consider age classes, sex ratios and distribution of the populations (DHRSG, 1993).

Furthermore the term ‘certainty’ has been addressed (DHRSG, 2004b). It is recognised that absolute certainty can never be obtained. Therefore, this term is defined as ‘reasonable certainty’, based on scientifically sound research. If no reasonable certainty is obtained regarding substantial effects, no exemption or dispensation can be granted.

Area protection BD and HD

In order to determine whether the provisions of the BD and HD for area protection (article 6 HD; Box 2) apply, it must be checked if an activity or a project will take place in or close to an area designated as SAC. Direct as well as external effects must be considered. External effects can arise on long distances (e.g. the development of harbours and industrial areas near Rotterdam in the river estuaries in the south-western part of the Netherlands affected the Wadden Sea in the northern part of the country; ABRvS 26 January 2005). If that is the case, then the possibility of significant effects must be assessed. This assessment must be related to the conservation and management objectives concerning the SAC (CoJEC 7 September 2004). When these objectives have not (yet) been formulated, the assessment must be related to qualifying habitats (HD Annex I) and/or species from HD Annex II and BD Annex I. Potential cumulative impact must also be evaluated. If significant effects are probable or cannot be excluded with certainty, an ‘appropriate assessment’ must be carried out. This assessment must determine whether there is certainty that no significant effects arise on the natural characteristics of the area (among other things suitability for qualifying species and natural habitats, and all species typical for that natural habitats and all habitats of qualifying species; Box 2). Also the accuracy, reliability and or probability of the predictions must be indicated (European Commission, 2002). The appropriate assessment must be based on best scientific knowledge. Mitigating measures may be considered in this assessment.

Box 2: Article 6 of the Habitats Directive and criteria for appropriate assessment (European Commission, 2002; Vos ***et al.***, 2002; Neumann & Woldendorp, 2003).

Article 6:

1. For special areas of conservation, Member States shall establish the necessary conservation measures involving, if need be, appropriate management plans specifically designed for the sites or integrated into other development plans, and appropriate statutory, administrative or contractual measures which correspond to the ecological requirements of the natural habitat types in Annex I and the species in Annex II present on the sites.

2. Member States shall take appropriate steps to avoid, in the special areas of conservation, the deterioration of natural habitats and the habitats of species as well as disturbance of the species for which the areas have been designated, in so far as such disturbance could be significant in relation to the objectives of this Directive.

3. Any plan or project not directly connected with or necessary to the management of the site but likely to have a significant effect thereon, either individually or in combination with other plans or projects, shall be subject to appropriate assessment of its implications for the site in view of the site's conservation objectives. In the light of the conclusions of the assessment of the implications for the site and subject to the provisions of paragraph 4, the competent national authorities shall agree to the plan or project only after having ascertained that it will not adversely affect the integrity of the site concerned and, if appropriate, after having obtained the opinion of the general public.

4. If, in spite of a negative assessment of the implications for the site and in the absence of alternative solutions, a plan or project must nevertheless be carried out for imperative reasons of overriding public interest, including those of a social or economic nature, the Member State shall take all compensatory measures necessary to ensure that the overall coherence of Natura 2000 is protected. It shall inform the Commission of the compensatory measures adopted. Where the site concerned hosts a priority natural habitat types and/or a priority species, the only considerations which may be raised are those relating to human health or public safety, to beneficial consequences of primary importance for the environment or, further to an opinion from the Commission, to other imperative reasons of overriding public interest.

Criteria for appropriate assessment (related to article 6.3 and 6.4):

- Expected developments must be assessed using the autonomous development and including planned or executed physical reconstruction and management measures.
- The conservation status for (the population of) each of the examined species must be assessed in terms of quantity (distribution and population size) and quality (vitality, reproduction and mortality) and ultimately the probability of species extinction. Accuracy and reliability of predictions must be indicated.
- The conservation status of the natural habitats must be assessed in terms of quantity (distribution and surface area) and abiotic quality (specific structure and functions).
- The assessment of habitat and species should take into account:
  - Direct and indirect, secondary and cumulative effects;
  - Effects on short term as well as long term;
  - Local effects as well as effects on the Natura-2000-network;
  - Management objectives of the protected areas and the objectives of the directives.
- Compensation: Qualitatively and quantitatively identical to lost values, related to qualifying species, natural habitat and corridor functions, and types of ecosystems.
- Recent data must be used.

If certainty has been obtained that no significant effects will take place, no continuation of the appropriate assessment is required. If not, it must be examined whether there are realistic alternatives for the activity to be carried out. This requires a broad perspective that may surpass the local governmental territory. Other projects must also be taken into account, based on the objectives of the competent authority. If there are no alternatives without significant effects, it must be motivated for which imperative reasons of overriding public interest, including those of a social or economic nature, the plan or project must nevertheless be carried out. This also requires a broad perspective. However, if only the BD is concerned, socio-economic reasons cannot be called upon. If imperative reasons of overriding public interest exist, all necessary compensating measures must be taken to guarantee that the coherence of Natura 2000 remains intact. Advise of the European Commission will be required if priority species or habitats are concerned.

3 Ecological framework

Ecology is often defined as the science which studies the interactions between organisms and their environment, and patterns and processes of ecosystems. It is one of the many biological disciplines. Biodiversity, the conservation of which is a central issue within the legal framework, may be assessed on three levels (genetic, species and ecosystem; UNEP, 1992). Each of the levels has four components (Noss, 1990): (1) composition: what is there and how much; (2) structure in space: spatial distribution of, and spatial relations between, for example, species and areas; (3) structure in time: seasonal and diurnal cycles, and (4) processes: physical, chemical and biological processes. In addition to ecology, many other biological disciplines are relevant for impact assessment and legal procedures, e.g. biogeography, population dynamics, population genetics and ecotoxicology (Smith, 1992; Bakker ***et al.***, 1995). Different disciplines in ecology are also distinguished based on the type of ecosystem they study, e.g. forest ecology and river ecology. In this paper we pay special attention to river ecology, in order to study the consequences for river management of the implementation of ecological knowledge in legal instruments.

3.1 Ecological approaches

Within ecology several approaches evolved. Ecology as a scientific discipline has arisen from natural history (Benson, 2000). This approach emphasises making inventories of nature and particular components. A next step was searching for patterns that indicate functional mechanisms on the level of ecosystems. With the rise of ecology as modern science in the early 20th century, the ecosystem approach became dominant. This approach aims at the development of deterministic explanatory models concerning structure and dynamics of ecosystems (cf. Odum, 1969). Within the systems approach two competing paradigm exist. In the so-called ‘nature in balance paradigm’, nature is conceived as a more or less deterministic, homeostatic, phenomenon that generally is in balance and in which scale is no critical variable (Pimm, 1991; Wiens, 1999). Over the last decades the so-called ‘dynamic equilibrium paradigm’ has arisen in ecology. Natural systems are conceived as open, non-deterministic and not in balance. Moreover, patterns and processes are considered to be highly dependent on spatial and temporal scale. Spatial heterogeneity is seen as very important for biodiversity patterns. Ecosystems are viewed as an evolutionary interplay of several factors and organisms which can interact in various ways (Sagoff, 2003). Moreover, non-linear and chaotic phenomena inherently yield high complexity and uncertainties of predictions concerning human impacts on species and ecosystems (Haila, 2002; Scheffer ***et al.***, 2001). However, uncertainties can be analysed and quantified (cf. Ragas, 2000; Geneletti ***et al.***, 2003).

Although the abovementioned paradigms consecutively evolved, they nowadays co-exist. In ecology, the ‘dynamic equilibrium paradigm’ has been firmly engrained, but also the ‘nature in balance paradigm’ is still very important. The ‘natural history paradigm’ is hardly used anymore in ecological science, but is still relevant in nature conservation (cf. Simberloff, 1998).

Concepts of river systems

In river science the 'dynamic equilibrium' paradigm, with the emphasis on spatial and temporal heterogeneity, became dominant in the last decades (Wiens, 1999; Leuven ***et al.***, 2002). River floodplain ecosystems are characterised by dynamics: hydrodynamics, morphodynamics and vegetation succession cycles. These dynamics cause high levels of spatial and temporal heterogeneity. On the scale of floodplain areas, inundations are disturbances that set back succession and population levels. On higher levels of scale, distribution of landscape ecological units (e.g. ecotopes) tends to remain constant over time (Ward ***et al.***, 2002).

Ecosystems, and especially river ecosystems, are governed by patterns and processes that are highly scale-dependent and non-deterministic, resulting in a spatially heterogeneous and non-equilibrial landscape (Ward ***et al.***, 2002).

Odum (1969) provided a strong theoretical basis for regarding ecosystems as hierarchically organised and governed by flows of energy and matter. The hierarchical nature of river systems was demonstrated by Frissel ***et al.*** (1986) and Townsend (1996). Pickett & White (1985) point out that ecosystems can be understood as composed of functional elements, linked by dynamics of water, earth, air and solar energy (Patch Dynamics Concept). The Hierarchical Patch Dynamics concept further conceptualises links between patches in the river catchment hierarchy across various temporal and spatial levels of scale (Poole, 2002).

Links between functional elements play important roles over gradients along the longitudinal axis (River Continuum Concept (Vannote ***et al.***, 1980), Nutrient Spiralling Concept (Webster & Patten, 1975; Newbold ***et al.***, 1981), the transversal axes of rivers (Flood-pulse concept; Junk, 1989) and the vertical axes of rivers (Townsend, 1996). The importance of the fluctuation of stream power to biodiversity patterns in both the lateral and transversal axis has been conceptualised by the Flow-Pulse concept (Tockner ***et al.***, 2000). The Serial Discontinuity concept models rivers whose natural dynamics have been suppressed by regulation (Ward & Stanford, 1995).

The Intermediate Disturbance hypothesis (Huston, 1979) predicts that biodiversity is highest with intermediate levels of disturbance, for example flooding by the river. Empirical evidence (Van den Brink ***et al.***, 1994, 1996; Ward & Tockner, 2001; Aarts ***et al.***, 2004; De Nooij ***et al.***, 2006b) shows that different taxonomic groups react differently.

Species selection

Ecological reasons for selection of protected species are a certain degree of extinction risk and the ecological function of a species. Species can be threatened, but not rare (House Sparrow) or rare but not threatened (King Fisher). Rarity may have natural or anthropogenic causes. Natural causes are adaptations to specific environment circumstances (e.g. the Zinc Violet that is endemic to metalliferous soils in East Belgium, South Netherlands and Western Germany) or a limited reproduction capacity (Eurasian Eagle Owl). Moreover, the bio-geographic context is relevant. For example, in a particular country a species can be rare because it exists on the edge of its natural distribution range. In the Netherlands, this is the case with a number of orchid species, and the Banded Fire Salamander. Anthropogenic causes for rarity frequently relate to habitat destruction (Spotted Owl) and/or hunting and traffic (European Otter). River characteristic species often have specific adaptations that enable them to survive in environments characterised by hydrodynamic disturbance (De Nooij ***et al.***, 2006b). This also means that these species have limited competition power outside riverine environments. In heavily modified river floodplain ecosystems, many species adapted to intermediate levels of disturbance, species adapted to high levels of spatial heterogeneity (Petts, 1989) and species sensitive to pollution have become rare (Leuven ***et al.***, 2005).

Each category of rare species is ecologically different (Rabinowitz ***et al.***, 1986). The causality between human activity and rarity greatly varies. The rarity and trend in number of populations and distribution areas of both the species and its habitat form the vital criteria for extinction risk. Moreover, quantitative and qualitative aspects of populations and habitats are relevant. Species with the highest extinction risk are rare species with genetically impoverished and/or small populations that exhibit a decreasing trend and are bound to a rare and decreasing habitat.

Ecological functions of species relate to food chains, ecosystem processes (for example biomass production, decomposition and succession) and structuring communities (e.g. vegetation structure). According to Simberloff (1998), species with a large influence are called key stone species. Species can also be indicative for a certain environmental quality or characteristic of a certain type of ecosystem. Moreover, species can have a so-called umbrella function. Their habitat requirements are so broad that they also cover the habitat requirements of a lot of other species. Protection of umbrella species may therefore result in the protection of many other species However, ecological scientists still heavily debate the umbrella and key stone species concepts as well as the identification of these types of species. Key-stone species in river floodplain ecosystems are species that have a large influence on vegetation succession and structure and on the distribution of water flow within the floodplain area. Examples are willow species that can grow into a dense willow bush, with a high resistance to water flow, within five years (Baptist ***et al.***, 2004), or beavers that can thin out willow bushes within the same period, and tend to build dams that alter the water levels in the floodplain area (Johnston & Naiman, 1990; Collen & Gibson, 2000).

3.2 Procedures and term definitions in ecology

The study of human impacts on species and ecosystems firstly requires a delineation of the area potentially affected, in space and time, that takes into account the physical, chemical and biological influences of the activities. Biological influences refer to, for example, introduction or capture of organisms. The known sensitivity of various taxonomic groups for environmental impact is also taken into consideration. Effects on the short and the long term, and the duration and continuity of the effects are important (Dutch Ministry of Spatial Planning, Housing and the Environment, 1983). Hereafter, the physical, chemical and biological characteristics of the area affected are determined. These concern the vegetation - and soil structure, hydromorphology, biogeochemistry, chemical composition of (ground) water and soil, controlling processes such as grazing, management, hydrodynamics and climatic circumstances, and present ecosystem types, phytosociological units and species assemblages.

For river systems especially the spatial and temporal distribution of hydrodynamics, and erosion and sedimentation processes are the most important physical characteristics to investigate (Junk ***et al.***, 1989). This can be done by combining elevation maps with hydrographs. Chemical features must be characterised in relation to the flood pulse and the distribution of chemicals between the various compartments of river systems (sediment, water, biota, and to a lesser extent air): the concept of nutrient spiralling (Newbold ***et al.***, 1981). Erosion and sedimentation processes and patterns are highly important for the distribution of pollutants in the floodplain area.

For the species present, the distribution in space and time of the local population is determined (including the population size and seasonal migration patterns). There are three organisation levels: organism, population and species. A population is a group of organisms of the same species which can potentially reproduce. For the population, three levels of scale are considered: the local population, the metapopulation and the world population (Bakker ***et al.***, 1995; Verboom ***et al.***, 2001).

Subsequently, the functional relations between the relevant species and its environment are analysed, involving the traits that enable the species to survive in a particular landscape. The term habitat refers to the whole of physical, chemical and biological factors in space and time that a species requires for completing its life cycle (Smith, 1992). For plants this concerns the so-called site factors (Bakker ***et al.***, 1995), which are important for establishment (germination) and persistence (development and survival). Animals frequently require several habitat patches which they use for different phases in their life cycle, such as reproduction and hibernation, and different circadian activities such as foraging, resting and migration between habitat patches. Habitat suitability analyses consider the surface area, connectivity, heterogeneity and configuration of habitat patches (Southwood, 1977; Smith, 1992; Verboom ***et al.***, 2001).

In river floodplain ecosystems, connectivity relates to the connectivity of various types of water bodies with the river, depending on the flood pulse. Especially heterogeneity of the riverine landscape in space and time is a highly important feature. As mentioned before, most river species have specific adaptations to hydrodynamic disturbance, shifting mosaics, and varying connectivity. Especially fauna species with limited migration capacity must invest in life strategies that enable them to survive in highly variable environments (Wijnhoven ***et al.***, 2005; 2006).

Considered characteristics concerning migration are the home range, migration time and migration routes of species involved and the required landscape elements (corridors and stepping stones). These characteristics can be very important for the delineation of the area affected (for example external effects on habitats can arise owing to deteriorations of corridors and stepping stones.

Each species has its own ecological requirements, including spatial and temporal patterns. This can even partially differ from area to area for the same species (European Commission, 2000). For this reason, the species specific habitat and corridor function must be determined for every particular area. In addition, interactions between species must be investigated (e.g. predation, competition, parasitic and symbiotic relations; Smith, 1992). Furthermore, the ecological range of a species must be involved, i.e. the tolerance limits for various environmental conditions. Generally, environmental specialists (narrow range) and generalists (broad range) are distinguished.

With the abovementioned information, the impact of an activity on protected species (disturbance) can be determined. Within the context of the mentioned Intermediate Disturbance Hypothesis, the term ‘disturbance’ is used for changes in the physical landscape. Disturbance in this sense is not necessarily a negative influence. When disturbance means that organisms must adapt to new environmental conditions, this decreases energy available for growth and reproduction (i.e. stress). Disturbance is also defined as the influence of physical, chemical and/or biological factors leading to decrease of growth, survival and/or reproduction rate (Bakker ***et al.***, 1995). Disturbance causes primary and secondary effects on the short term as well as the long term and on various spatial scales. Both the ecological demands and ecological ranges of a species determine to what extent the environment can change before negative effects on individuals and population arise (ecological thresholds).

Exceeding of ecological thresholds for habitat quantity and habitat quality results in increase of mortality and decrease of natality. Mobile species might also migrate to other areas. In addition to sensitivity of species, also sensitivity of its habitat for environmental change can be very relevant. It is possible that a habitat already becomes unsuitable due to relatively small environmental changes. Cumulative effects in space and time can arise for both species and their habitat (Leuven ***et al.***, 1998; 2002).

Different organisms may show divergent stress responses to environmental disturbance. Moreover, the response depends on the health condition of the organism (e.g. sickness can influence stress tolerance). This is relevant for cumulative effects. Conversely, the ecological range of organisms can change as a result of habituation and genetic adaptation to environmental stressors. Many plants and animal species are evolutionary adapted to specific disturbances, e.g. a recurrent flood pulse in riverine ecosystems.

When disturbance leads to a decrease of population size, the extinction risk of the local population must be assessed. Extinction risk is influenced by recovery capacity through, for example, reproduction or recolonisation. Populations are characterised by dynamics. Variables for population dynamics are birth rate, mortality, immigration and emigration of individuals, age structure, sex ratio and generation time. Furthermore, the genetic diversity of the population is important for extinction risk assessment. These characteristics of populations are also required for determination of the minimum viable population size and recovery capacity after disturbance (Bakker ***et al.***, 1995). When the decrease of numbers exceeds the natural fluctuations, the extinction risk of the local population is high. Furthermore, decrease of numbers may lead to decrease of the viability of the population. In order to assess negative effects on population viability, also the relative importance of affected individuals – age, reproductive capacity – must be taken into account. Genetic impoverishment is often caused by decrease of exchange of genes (e.g. inbreeding) and leads to loss of adaptive and reproductive capacity, which may lead to a downward spiral in population vitality (Primack, 1993). On the basis of information on the local population, the measures required for mitigation of effects can be determined. When there is a heightened risk of extinction or genetic impoverishment of the local populations, the consequences for the metapopulation must be assessed as well.

A metapopulation is a spatially structured population, divided in subpopulations that occur in habitat patches that form a network from the viewpoint of the species (Bergers & Opdam, 1996). A local population often does not stand alone. Therefore, recovery can also take place from another subpopulation, if these two subpopulations are connected. Conversely, the metapopulation can be negatively affected when one of its subpopulations is weakened or disappears. Very rare species often have local populations that do stand alone, and that are therefore extra vulnerable. For this reason, it must always be assessed to what extent there is a functional metapopulation. The migration capacity of a species determines the spatial scale on which the metapopulation must be investigated. Furthermore, information on the relative importance of the affected local population and its relation with other subpopulations is required. On the basis of this information the recovery capacity of a local population by migration or dispersal from another subpopulation can be evaluated. Moreover, the effects of a human activity on metapopulations can be assessed for example in terms of the expected decrease of numbers of populations and the risk of genetic impoverishment. After this analysis, the measures required for compensation of the disturbance of a local population can be determined, taking the metapopulation into account. Species with low population dynamics, limited dispersal capacity and/or a specific life history/strategy have the highest extinction risk. When the metapopulation is threatened, the extinction risk of the world population must be assessed.

4 Comparison of the ecological and the legal framework

Appendix 1 provides an overview of the aspects relevant within the ecological framework as described in paragraph 3, and the implementation of these aspects in legislation and jurisprudence. For legislation, the quality of this implementation is indicated on a scale ranging from correctly implemented, to not implemented at all. For jurisprudence, the evaluation is done in terms of how often an ecological aspect is mentioned and whether ecological aspects are treated correctly. This paragraph focuses on aspects of legislation that are, according to ecological criteria, not correctly implemented, i.e. too wide or too narrow for meaningful ecological effect assessment. For jurisprudence, the focus is on missing ecological aspects, and on aspects that are missing in legislation, but are included in jurisprudence.

4.1 Which aspects of legislation are too wide for ecological science?

Within the legal framework species are selected based on ecological, societal, ethical and aesthetic criteria. The recognition of intrinsic value in the FFA leads to legal protection of individual organisms, whereas in ecological science individual organisms are not the central issue. Stressing of individuals can hardly be measured without controlled laboratory conditions and in many cases it is even impossible to predict the consequences of stress for individuals, let alone populations.

Concerning species protection, the criterion to conserve species in their natural distribution range in a favourable state of conservation means that an activity that causes extinction of a local population and a regional metapopulation without significant effects on the favourable conservation status of the species population as a whole (i.e. global population), would be allowed.

Ecology cannot determine cultural, economic and recreational demands to population levels as mentioned in the aims concerning legal area protection. These societal issues do not belong to ecological science. Also the demand for certainty about environmental effects cannot be met by ecological science, there is always some uncertainty. We argue that the demand for certainty results from a limited vision on nature in the legislation, i.e. nature being viewed as a deterministic phenomenon that generally is in balance. Another aspect of legislation concerning compensation that leads to information demands that are too wide for ecology is the demand that compensation of effects must ensure identical surface area and quality. These criteria are also problematic, because strictly speaking, natural characteristics of areas cannot be replaced in an identical way, and if they could, ecological science would not be able to determine if compensation resulted in an identical situation. Nevertheless, the criteria for compensation do provide a guarantee that compensation measures are taken seriously. In order to function, these criteria must be used in a flexible way. In the next paragraph opportunities are given from an ecological point of view.

4.2 Which aspects of legislation are too narrow for ecological science?

Approach to nature and species selection

Because of the large attention to particular components, for example (rare) species, the legislation seems to be based on the oldest paradigm in ecology (i.e. natural history). In addition the ‘nature in balance’ paradigm has had a large influence. This can be deduced from the fact that legislation emphasises conservation and certainty concerning consequences of an activity on protected species. The influence of the ‘nature in balance’ paradigm is also indicated by the limited attention paid to dynamics, heterogeneity and spatial and temporal scale levels of populations and ecosystem functioning. Of the three levels (genes, species and ecosystems) and components (composition, structure in time and space and processes) of biodiversity, particularly the genetics and processes are underrepresented in the legal framework. In many cases, however, genetic properties and/or processes (e.g. key processes such as gene flow) can be decisive (Slootweg & Kolhoff, 2003).

Area protection is conservative by nature, whereas ecosystems are characterised by succession and rejuvenation, dynamic equilibriums and a certain degree of stochasticy. Natural dynamics as essential - and potentially compensating or worsening - process is frequently overlooked. This is particularly problematic within the framework of projects aiming at ecological rehabilitation and flood defence in riverine landscapes, which are characterised by rehabilitation of dynamics such as flooding, grazing and succession. Within the context of species protection, important positive or negative effects of dynamics might be overlooked.

Although also in ecology much emphasis still lies on rare species, the protection of particular species without their ecological context (causal reasons for rarity, ecological function) is difficult to handle. Conversely, jurists have large problems with the importance of ecological context and scale dependence, which can remarkably differ per species and area.

Contrary to legislation, ecology does not a priori restrict the selection of species, life history traits and environmental variables. Moreover, species selection in the legal framework largely ignores the ecological significance of species and their extinction risk. This results in a lack of attention to plants, invertebrates and micro-organisms, whereas these taxa represent the largest biodiversity and undisputedly are very important for structure and functioning of ecosystems. Figure 1 shows the unbalance between globally threatened species occurring in the European Union and protection status of these species.

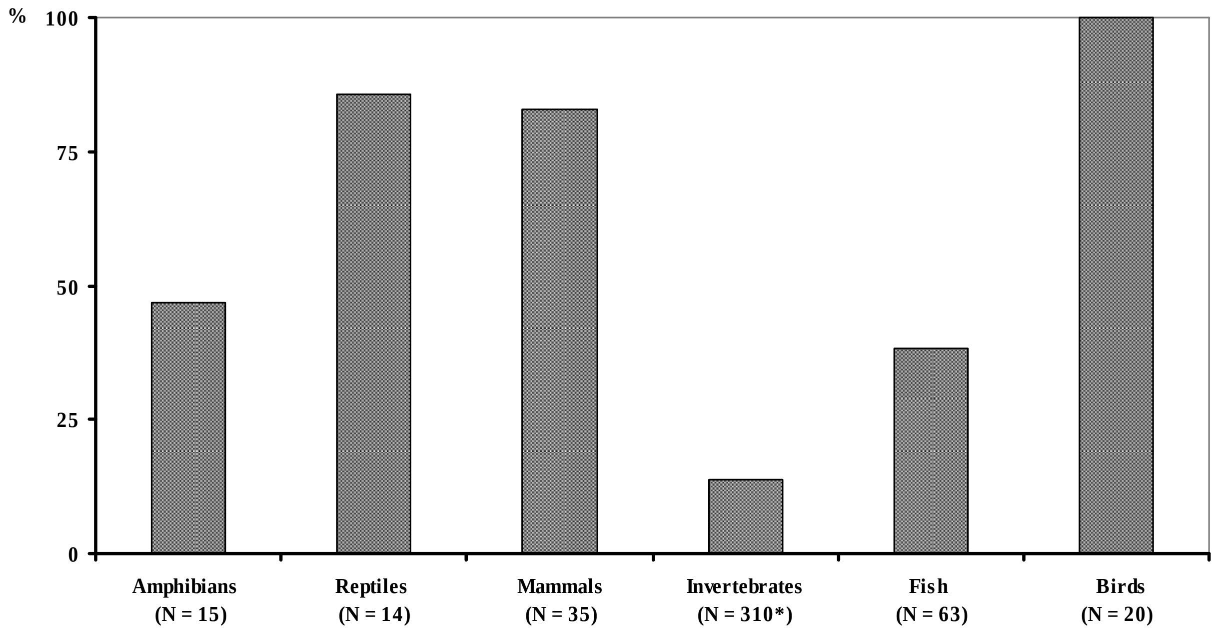

Figure 1. Percentage of inclusion of globally threatened species occurring in the European Union in protected species lists of EU directives and the Bern Convention (Data: European Environment Agency, 2005). N: Number of globally threatened species occurring in the European Union;
*: Only crustaceans, insects and molluscs were considered.

For the Netherlands the pattern is largely similar. Table 1 shows that endangered higher plants, fish and invertebrates are largely unprotected, whereas birds are overrepresented. This also holds for river characteristic species.

Table 1. Numbers of red-listed and protected (river) species in the Netherlands.

Taxon	Red-listed species	Protected species FFA	Red-listed river species	Protected river species FFA
Higher plants	499	104	136	25
Birds	57*	697**	27	73
Reptiles and Amphibians	15	23	6	7
Mammals	21	66	6	9
Fish	24	12	19	6
Butterflies	47	26	17	9
Dragon- and Damselflies	27	8	6	3
All groups	690	936	217	132

*: breeding birds only; **: all species indigenous to Europe are protected in the Netherlands; FFA: Dutch Flora and Fauna Act.

Procedures and terminology

Appendix 1 presents the implementation of ecological aspects in the legal procedures. Concerning characterisation of effects and effect area, species protection includes only information on presence of species. Area protection includes qualitative and quantitative aspects of environmental effects and many characteristics of affected areas, albeit in very general terms.

Characterisation of species, populations and habitat patches in the legal framework ignores genetic diversity, causal reasons for decline and ecological function of species, and temporal aspects of the presence of populations. Area protection includes rarity and trend of habitats of species, whereas species protection does not.

The legislation only mentions particular parts of habitat. In the BD and HD habitat is defined as an environment in which a species lives during one of the phases of its life cycle. In the FFA definition, parts of habitat are neglected: for example migration habitat and foraging habitat. Furthermore, habitat quality, spatio-temporal requirements (spatial scale, habitat connectivity, configuration, migration), local relationships and species specific adaptations are not considered in both area and species protection.

Concerning effect assessment, area protection includes all kinds of environmental change. However, this is implemented incompletely and in very general terms (change of ‘natural characteristics of the area’). Species protection ignores environmental change as such, but is focussed on effects on habitat quality and quantity. Sensitivity of habitat, including corridors of species, and qualitative aspects of corridors are ignored in the legal framework. Assessment of effects on individual organisms, relevant only within the context of species protection, mentions only disturbance of individuals on the short term (i.e. alarming).

The terms disturbance and alarming as used in the FFA strongly deviate from ecological approaches. In the legal framework disturbance is by definition a negative influence, whereas in ecology disturbance is a result of environmental dynamics that can be beneficial (disturbance conditions are around the ecological optimum) as well as detrimental to species (disturbance conditions are close to, or exceed, tolerance limits). In the latter case, disturbance leads to stress, which is, by many ecologists, defined as reduced survival probability.

Species protection distinguishes between direct influence of human activities on individuals (alarming) and indirect influence on species via habitat changes (disturbance). However, this difference can be hardly made operational in ecological effect assessments. In river systems disturbance of the physical landscape is inherent, and does not always have to be regarded as a negative influence. Therefore in these systems anthropogenic influence on existing disturbance patterns must be examined. The prohibition provisions of FFA (art. 11) certainly have ecological significance, but they only concern parts of habitat and habitats of plants are not protected. Protection of plants is therefore particularly inappropriate.

The definition of the term metapopulation in the FFA is badly chosen. In the FFA the metapopulation seems to mean the entire global population, instead of a spatially structured population, divided in subpopulations that occur in habitat patches that form a network from the viewpoint of the species (Bergers & Opdam, 1996).

Current legislation pays no attention to genetic variation within species, interaction with other species, population structure and dynamics, viability of populations, minimum viable (meta)population size, disturbance on the long term, influence of condition or sicknesses on tolerance levels and behavioural response of animals to disturbance. This means that important negative effects on protected species may be left out of consideration in legal procedures.

In daily practice of area protection, the poor implementation of abovementioned ecological terms leads to a limited view on compensation of negative impact on ecological networks. Furthermore, the demand of identical quality and area is problematic, as stated in the last paragraph. When interpreted in a rigid fashion it strongly limits practical opportunities for compensation. Furthermore, rigid interpretation is in many cases not necessary from an ecological point of view. The demand is probably intended to guarantee that lost natural values are replaced with something that is equal to its ecological function (i.e. function for qualifying species and the coherence Natura 2000). According to ecological criteria, alternative ecosystems at the same location or similar ecosystems with smaller surface area but higher quality may yield higher ecological values or improve the ecological network for the species concerned (e.g. Natura 2000).

The poor implementation of ecological networks, metapopulations and population dynamics leads to difficulties in making operational the term ‘favourable conservation status’. In many cases, assessment of effects on individuals is ecologically irrelevant. However, assessment of effects on the species level (i.e. world population) is too crude. The most appropriate level is that of the regional population and the ecological network (Broekmeyer ***et al.***, 2003).

4.3 Analysis of jurisprudence

General analysis

In the Dutch jurisprudence analysed, many ecological characteristics of species, habitats and effect areas were not mentioned (appendix 1). This was especially the case when jurisprudence concerned species protection. Characteristics of the protected species, habitats and areas concerned that were not included mainly relate to bio-geographical and genetic aspects, the relevant spatial and temporal scales, and the dynamic and heterogeneity aspects of habitats. Effect assessments in most cases neglect chemical aspects, soil characteristics and ecosystem processes, the sensitivity of the habitats concerned, the vitality of the population and individuals, species behaviour, interaction between species, population dynamics, population genetics and metapopulation aspects.

Characteristics of species and areas that are missing in legislation are sometimes included in jurisprudence (appendix 1). Examples are spatio-temporal aspects of effects and habitat requirements, local relationships, tolerance levels and natural fluctuations of populations. Ecological characteristics mentioned sporadically in Dutch jurisprudence are for instance acreage and connectivity of species’ habitats (ABRvS 18 January 1999; Rb Alkmaar 4 June 2004). Sometimes, verdicts also refer to corridor function and migration of species (Rb Alkmaar 4 June 2004; Rb Haarlem 21 April 2004), tolerance levels of species (Rb

Leeuwarden 23 January 2004) and metapopulation aspects (Rb Alkmaar 4 June 2004). Jurisprudence regarding river floodplains in the Netherlands has included dynamics of sediments and habitat types (ABRvS 16 July 2003). In addition, other jurisprudence mentioned habitat heterogeneity (Rb Alkmaar 4 June 2004) and uncertainty (ABRvS 5 September 2003). In the rest of this paragraph, special attention is paid to jurisprudence that concerned floodplains in the Netherlands.

Wind turbines in the IJssel Valley (ABRvS 17 December 2003)
In this case the construction of wind turbines close to a protected area (SAC) under the Birds Directive, the riverine area called IJsselvallei (IJssel Valley), was suspected to have negative effects on the populations of several bird species (Geese and Swans) for which the IJssel Valley had been designated. Because the turbines were planned in the hart of the flying corridor of these bird species, they might cause population decrease. Furthermore, disturbance of resting and breeding areas of various grassland bird species was expected.

The effect assessment was based on flying intensity and flying routes, location of breeding sites and (potential) foraging areas, visibility of the turbines at night and the placing of the turbines parallel to flying routes. The assessment resulted in estimated percentages of population decrease, which were below 5%. This was considered to be insignificant. The judge decided that the activity could be carried out as planned.

Harbour in the Wageningen Floodplains (ABRvS 21 January 2004)
The expansion of a harbour in Wageningen was expected to have negative effects on a SAC designated under the Birds Directive. These effects could be caused by increase of noise levels, which could disturb several qualifying bird species. Furthermore, effects on a SAC designated under the Habitats Directive (i.e. the Veluwe) might arise. Two amphibian species protected by the habitats directive, the Natterjack toad and Crested Newt, both present on Annex IV might suffer from the plans.

The effect assessment concerning the area protected by the Birds Directive was based on the spatial distribution and intensity of noise in relation to use of the area by the bird species. In addition, habituation of the birds to the noise was taken into account. The highest intensity of noise was predicted to arise in a part of the area that is relatively dry, and that had hardly any function as habitat for bird species except for the White-fronted Goose. The effects were considered not significant. The effect assessment concerning the area protected by the Habitats Directive was based on the distance of this area from the planning area (two kilometres). No significant effects were expected.

Effect assessment concerning the two protected amphibian species was based on use of the area as terrestrial habitat by these two species. Available data did not point out that the area was used. Therefore, no significant effects were expected. The judge decided that the activity could be carried out as planned.

Container terminal in the Westerschelde Estuary (ABRvS 16 July 2003)
The construction of a container terminal on the banks of an estuary might cause effects on a SAC designated under the Birds Directive as well as the Habitats Directive.

Effect assessment was based on the fact that within the planning area. which is only a small part of the SAC, the natural features would completely disappear, leading to destruction of the habitats of bird species as well as natural habitats (HD I) for which the area was designated. Therefore, it was studied whether the effects of proposed compensation measures would be sufficient. Aspects taken into account were the natural habitat types (i.e. ecotopes) that were lost and their surface area, the importance of newly created natural habitats relative to the actual situation, the autonomous development of natural habitat types, the rarity of natural habitats, the function of natural habitat types for coastal bird species as breeding and foraging habitat, high-water-free refuge place, and as resting, foraging and winter habitat for migratory bird species. Habitat types that were lost were compensated partly by different habitats, which were supposed to be more effective than compensation by creation of the same type of habitat. The judge decided that the activity could not proceed as planned, because of insufficient investigation of possible alternatives for the construction of the container terminal.

Evaluation with respect to riverine aspects
From the analyses above it becomes clear that the dynamic aspects of riverine areas (inundation, erosion and sedimentation) are as yet not considered very important in jurisprudence. One reference is made to soil humidity, without referring to inundation or river water levels (ABRvS 21 January 2004). The use of terrestrial biotopes by amphibians is mentioned in one case, but not related to the importance of these biotopes as refuge places when the area is flooded (ABRvS 21 January 2004). However, in another case (ABRvS 16 July 2003) high-water-free refuge places are mentioned, which implies awareness of fluctuating water levels. In this latter case the dynamics of a habitat type are mentioned in terms of the autonomous development of the surface area of this habitat type, as related to erosion and sedimentation processes.

5 Conclusions and recommendations

5.1 Implementation of ecological knowledge in legal instruments

The legal framework aims at conservation and restoration of wild flora and fauna, which is also described as insuring biodiversity. Species are selected based on ecological, societal, ethical as well as aesthetic criteria. Prohibition provisions of species protection (FFA) forbid removal of protected plants, and disturbance of individual animals and parts of their habitat. Dispensation can be obtained when offence of the provisions does not lead to negative effects of species populations. Areas protection (BHD) gives regulations that must insure maintenance of quantity and quality of protected area in terms of their ecological function for (populations of) qualifying species and natural habitats (i.e., phyto-sociological units), and their contribution to the coherence of the European ecological network Natura 2000.

5.2 Differences between the ecological and the legal framework

Aims and criteria for valid claims
Ecology aims at understanding patterns and processes in the biosphere, relying on empirical research and statistics (quantitative methods). Legislation aims at regulating human behaviour in order to conserve nature. Criteria for making valid claims are firstly logical consistency and feasibility in terms of providing clear procedures (qualitative methods). Ecological science does not aim at normative assessments of anthropogenic impacts (e.g. whether or not effects are significant in terms of legal criteria). Therefore, facts and figures of ecological research can at the most support rational normative decisions.

Approaches to nature, species selection and ecological terms
The legal framework appears to view nature as a deterministic phenomenon that generally is in balance. The latest paradigm within ecology conceives natural systems as non-deterministic and not in balance, but in flux, with dynamic equilibriums. Ecological criteria for ecological

selection of species are extinction risk, ecological function, whereas in legislation more criteria are applied.

The aims of the nature conservation legislation are so wide that, with the current body of ecological knowledge, the information requirements of various legal procedures cannot be met. There is lack of insight in the response of ecosystems, species and populations to various types of human activities, and there are large uncertainties in predictions. Effects assessments require prediction of physical and chemical changes and their propagation in ecosystems. This information can only be obtained when physical, chemical and ecological models are integrated, which still is a challenge to interdisciplinary science.

Ecological reality is much more complex than the legislator has implemented in legal instruments for nature conservation and several recent ecological insights have not yet or insufficiently been implemented. This causes not only too wide information needs, but also many too restricted criteria for determining negative effects. Species selections and legal protection formulas frequently are ineffective and too limited. Many ecological terms are not correctly implemented. These limitations of the legal framework may be related to criteria such as legal security, providing clear frameworks to maintain law and order, and internal consistency of the law. Moreover, in ecological science there still is much discussion about the ecological functions of species, extinction risks and term definitions. Ecological science can never provide answers to all the questions that arise in legal procedures, because: (1) they may be beyond ecology; (2) knowledge and information is lacking or impossible to gather in practice; and (3) fundamental uncertainties always remain.

Consequences for river management

Current river management strategies aim at increasing river dynamics in floodplains and combine flood risk reduction with nature development. This approach collides with the philosophy of nature legislation, which is oriented towards conservation. For the management of river floodplain ecosystems, ecosystem and landscape dynamics, heterogeneity and non-linear behaviour are crucial aspects. In addition, uncertainty regarding effect prediction plays an important role. Therefore the minor attention for these aspects in legislation can be problematic. However, within the legal framework there are various opportunities for including these aspects.

5.3 Improvement of the relation between the ecological and the legal framework

Legal framework

According to ecological criteria, integrated protection is required: species and their habitats, structures, abiotic conditions, processes and networks. The legal framework for nature conservation must be adjusted more in that direction.

Within a legal context, a restricted species selection is necessary. Legislation can not implement all potentially relevant ecological aspects of all ecosystems, and all 44,000 known species in the Netherlands. This would result in a very complex, even more information demanding, legal system which would offer low legal security and very limited opportunities to maintain the law. Moreover, in ecological science there is a lot of discussion about extinction risks and ecological functions of species. Therefore, the relation between ecology and legislation will never be in perfect harmony.

For improvement, priority should be given to the selection of species and areas on the basis of their ecological relevance and extinction probability, a proper implementation of species habitats, tolerance levels and (meta)populations in prohibition and dispensation provisions. Furthermore, legislation should give thorough attention to uncertainty, ecological networks and ecological relations in time and space. The HD already offers opportunities for protection of ecological networks through Natura 2000. Criteria for appropriate assessment of effects on SACs, state that accuracy and reliability of predictions must be indicated (European Commission 2002). The FFA explicitly recognizes that absolute certainty can never be obtained, and that therefore 'reasonable certainty' is enough. The legal framework therefore offers various opportunities for improvement.

Ecological knowledge that cannot be implemented in legislation might be included in legal management plans, codes of conduct, jurisprudence, and quality standards for ecological effect assessment. Important options for limiting negative consequences of differences between the two frameworks are:

1. Drawing up management plans for protected areas (SACs) that include relevant compositional, structural and functional aspects. The HD states that appropriate management plans can be designed for protected areas which correspond to the ecological requirements of the natural habitat types in annex I and the species in annex II present in the areas. Therefore, dynamics such as inundation and succession, and landscape heterogeneity, can become part of the conservation and management objectives concerning the area.
2. Ensuring that the codes of conduct within the framework of the FFA are well underpinned by ecological knowledge and include crucial life-history traits of species: habitat aspects, tolerance levels and (meta)population structure and functioning.
3. Paying more attention within jurisprudence to controlling (key) processes (environmental dynamics, succession and exchange of genes), functioning of ecological networks, and life history traits. Various verdicts already pay attention to acreage and connectivity of species' habitats, corridor function and migration of species, tolerance levels of species and metapopulation aspects. Jurisprudence regarding river floodplains in the Netherlands has already included dynamics of sediments and habitat types, habitat heterogeneity and uncertainty.
4. Providing quality standards for ecological effect assessment. These quality standards could include crucial species specific environmental factors and response variables. What does the species depend on, what are its tolerance limits and what is it capable of? Effect assessment must take place at the most relevant level, which is usually the regional metapopulation network. Assessment of effects on habitats of species must include foraging and migration habitat, spatio-temporal aspects and cumulative aspects. Population studies should pay attention to population structure and dynamics. Clear procedures for dealing with uncertainty in effect assessments must be designed, in which a certain degree of stochasticity is recognised as real and quantifiable phenomenon.

Ecological framework

Regarding the ecological framework, more insight is required concerning the distribution of protected species and important species they depend on, their habitat (the relation between organism and environment), and the response of these species and ecosystems to human activities. This body of knowledge must be developed by means of empirical and model research and could be disseminated by means of easy accessible (web-based) databases. Cumulative impacts should be studied more thoroughly. Furthermore, an experimental approach to impact studies could be adopted, in which for several activities (1) the initial situation is determined, (2) the developments after human interventions are monitored and compared to developments in reference areas so that causality and dose-response relations can be analysed, and 3) insight is gained in possibilities for extrapolation of known dose-response relations to other species, areas and types of measures. More insight in chaotic and non-linear phenomena is essential. Procedures for quantifying and dealing with uncertainties in an ecological as well as juridical context must be developed. Finally, species specific standards for the conservation of metapopulations must be determined (e.g. required probability of metapopulation viability over a period of 100 years; cf. Verboom ***et al.***, 2001). This calls for more financial resources (e.g. from project initiators) and improved coordination of research efforts related to environmental impact assessments (e.g. concerning flora and fauna surveys and the development and validation of predictive tools).

Attitude of professionals
The jurists and decision makers who apply the current nature conservation legislation should try and understand more of an ecological approach to nature conservation. Before criticizing legislation, ecologists should realise that the legal framework represents a world of thought in its own right, with its own objectives and criteria for making valid claims.

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Appendix 1. Ecological characteristics of effects, species and ecosystems mentioned in legislation and jurisprudence.

Ecological framework	**Legal framework** Legislation Species (FFA)	Areas (BHD)	Jurisprudence Species	Areas
Effect Characteristics				
Type and intensity of effects:				
- Change in (a)biotic conditions (vegetation structure, soil species, soil quality, hydrology, hydrochemistry, controlling processes, tremors, noise, light, heat, radiation, biota) and interactions between effects	-	*	*	***
- Environmental destruction (decrease of surface area)	-	***	**	***
- Environmental change (decrease of quality)	-	***	**	***
Spatio-temporal distribution:				
- Geographical distribution of effects	-	-	**	***
- Temporal distribution of effects (when and frequency)	-	-	**	***
- Duration (temporary-permanent)	-	-	**	***
- Degree of certainty of location, timing and way of execution.	-	-	-	*
Characteristics of effect area				
Biological (composition):				
- Species	**	**	***	***
- Phyto-sociological units	-	**	*	**
- Ecotopes	-	-	*	**
Physico-chemical:				
- Hydrogeochemistry	-	** n.e.	-	*
- Biogeochemistry	-	** n.e.	-	-
- Hydrology	-	** n.e.	-	**
- Soil structure (spatial structure)	-	** n.e.	-	-
- Vegetation structure (spatial structure)	-	** n.e.	**	***
Dynamics:				
- Landscape dynamics (succession, management, hydrodynamics, grazing etc., structure in time)	-	** n.e.	* n.c.	**
- Ecosystem processes	-	** n.e.	-	* n.c.
Characteristics of species, populations and habitat patches				
Ecological status:				
- Rarity	**	**	**	*
- Trend	**	**	*	*
- Biogeography (regional, continental, global)	**	**	* n.c.	* n.c.
- Rarity of habitat of species	-	***	*	*
- Trend of habitat (acreage, quality) of species	-	***	-	*
- Genetic diversity of species	-	-	-	-
- Cause of rarity/trend	-	-	-	-
- Keystone species (yes/no)	* n.e.	-	-	-
- Indicator species (yes/no)	-	-	-	-
- Umbrella species (yes/no)	-	-	-	-
Local population:				
- Geographical distribution	-	***	**	**
- Number of individuals/acreage	-	***	**	**
- Permanent/temporary	-	-	**	**
- Period (e.g. winter, during migration)	-	-	**	**
- Vitality/quality	-	-	*	*
- Genetic diversity	-	-	-	-

	Legislation		Jurisprudence	
Habitat, migration and spatial requirements (life history traits):	Species (FFA)	Areas (BHD)	Species	Areas
- Habitat and use of habitat in different stages of the life-cycle: combinations of physical, chemical, biological (including interspecies relations) factors	**	**	** n.c.	*** n.c.
- Habitat and use of habitat for different daily activities: combinations of physical, chemical, biological (including interspecies relations) factors	**	**	** n.c.	*** n.c.
- Spatial scale of habitat use for daily activities	-	-	-	**
- Routing in landscape for daily activities	-	-	*	**
- Timing of landscape use for daily activities	-	-	-	**
- Habitat specialism, -generalism	-	-	* n.c.	*
- Food requirements	-	-	*	**
- Migration capacity and spatial scale (in different stages of the life cycle)	-	-	-	**
- Migration habitat in different stages of the life cycle: combinations of physical, chemical, biological factors	-	-	-	**
- Migration time	-	-	-	**
- Migration routes	-	-	-	**
Spatio-temporal habitat demands:				
- Acreage	-	-	** [1]	**
- Configuration	-	-	*	**
- Connectivity	-	-	* [1]	*
- Heterogeneity (landscape diversity)	-	-	-	*
- Temporal habitat demands (temporal structure, dynamics)	-	-	-	-
Function of local area for species:				
- Habitat use	-	-	***	***
- Corridor use	-	-	* [2]	**
- Local relationships with other species	-	-	* [5]	*
Additional:				
- Survival skills e.g. specific adaptation	-	-	-	-
- When necessary: adjustment of effect area	-	-	*	*
Effect assessment				
Environmental change:				
- Vegetation structure	-	** n.e.	**	**
- Soil structure	-	** n.e.	-	-
- Landscape dynamics	-	** n.e.	*	**
- Hydrology	-	** n.e.	*	**
- Hydro-geochemistry	-	** n.e.	-	*
- Bio-geochemistry	-	** n.e.	-	-
- Ecosystem processes	-	** n.e.	-	*
- Cumulative effects	-	***	*	**
Effects on habitat:				
- Sensitivity of habitat including corridors of the species to environmental changes (physical, chemical, biological)	-	-	-	-
- Habitat quantity change (acreage)	**	***	**	***
- Corridor quantity change (acreage and connectivity)	-	**	**	**
- Habitat quality change (suitability)	**	***	**	***
- Corridor quality change (suitability)	-	-	**	**
Effects on individual organisms:				
- Tolerance levels (factors, levels, physical, chemical, biological)	-	-	** n.c. [3]	**
- Influence of diseases on tolerance levels	-	-	-	-
- Disturbance of individuals short term	***	n.a.	***	**
- Behavioural response to disturbance	-	-	-	*
- Disturbance of individuals long term	-	n.a.	*	**
- Adaptation to effects, increase of tolerance	-	-	-	*
- Cumulative effects in space and time	-	***	*	***

	Legislation		Jurisprudence	
Effects on local population:	Species (FFA)	Areas (BHD)	Species	Areas
- Increase of mortality	***	-	**	**
- Decrease of reproduction	-	-	-	-
- Avoidance by migration	-	-	-	-
- Natural fluctuations	-	-	*	**
- r- or K-strategy	-	-	-	-
- Mortality and Birth rate	-	-	-	-
- Generation time	-	-	-	-
- Recovery capacity by reproduction	** n.e.	-	-	*
- Influence of other species (predation/competition/mutualism/parasitism/commensalism)	-	-	*	*
Population composition:				
- Age	*	-	-	-
- Size	*	-	-	-
- Sex-ratio	*	-	-	-
- Numerical limits for population composition required for healthy population (number of reproductive units)	-	-	-	**
- Relative importance of affected individuals i.r.t. population (age, reproduction capacity, etc.)	-	-	-	-
- Effects on genetic diversity of local population (probability of genetic erosion/inbreeding)	-	-	-	-
- Effects on vitality	-	-	-	-
- Cumulative effects in space and time	-	***	-	
Expected decrease of local population:				
- Short term	**	***	**	***
- Long term	**	***	-	*
- Probability of extinction of local population	-	***	-	* [6]
Effects on metapopulation:				
- Geographical distribution (regional, national, continental & global)	**	-	-	-
Relative importance of effect area:				
- i.r.t. distribution area (central, sub-central, marginal)	**	** n.e.	-	-
- i.r.t. genetic diversity	-	-	-	-
- i.r.t. functioning of metapopulation	**	** n.e.	* [4]	-
Metapopulation characteristics:				
- Spatial scale relevant for metapopulation (dispersal capacity)	-	-	-	-
- Number of sub-populations linked to local population	**	-	-	-
Relation of local population to other sub-populations:				
- Degree of connection	-	-	-	-
- Direction of gene-flow (Source-Sink)	-	-	-	-
- Importance of local population for meta-population	**	** n.e.	*	*
- Recovery capacity by migration	** n.e.	-	*	*
Effects of activity on metapopulation:				
- Fragmentation	-	-	-	-
- Expected decrease number of sub-populations	-	-	-	-
- Expected decrease of genetic exchange	-	-	-	-
- Expected decrease of vitality of subpopulations including genetic erosion/inbreeding	-	-	-	-
- Cumulative effects in space and time	-	***	-	-
- Probability of extinction of the metapopulation (regionally, national, European, global)	-	**	-	-

Required mitigation and-or compensation	Legislation Species (FFA)	Areas (BHD)	Jurisprudence Species	Areas
Preventing significant effects on local population by mitigation:				
- Habitat acreage required	n.a.	***	n.a.	***
- Habitat quality required	n.a.	***	n.a.	***
- Habitat connectivity	n.a.	-	n.a.	**
- Maximum level of disturbance (physical, chemical, biological factors)	n.a.	-	n.a.	**
Compensating significant effects by creation of new habitat:				
- Coherence of metapopulation	n.a.	**	n.a.	*
- Optimal location of new habitat	n.a.	-	n.a.	**
- Habitat acreage required	n.a.	***	n.a.	***
- Habitat quality required	n.a.	***	n.a.	***
- Habitat connectivity	n.a.	-	n.a.	**

BHD: European Birds and Habitats Directives; FFA: Dutch Flora and Fauna Act.
Analysis of legislation: - = missing, *= mentioned, but not implemented in procedures, **= implemented in procedures, but incompletely, ***= OK, n.e. = not explicitly but in very general terms, n.a. = not applicable.
Analysis of jurisprudence: - = missing, *= mentioned sporadically, **= mentioned occasionally, ***= standard item, n.e. = not explicitly, n.c. = not completely, n.a. = not applicable.
1: In ABRvS 18 January 1999 AB 1999/357 and Rb Alkmaar 4 June 2004 LJN: AP1743 acreage and connectivity of species' habitat are mentioned.
2: In Rb Alkmaar 4 June 2004 LJN: AP1743 and Rb Haarlem 21 April 2004 LJN: AO 8078 reference is made to corridor function of the area and species migration.
3: In Rb Leeuwarden 23 January 2004 LJN: AO2334 tolerance levels of species are mentioned.
4: In Rb Alkmaar 4 June 2004 LJN: AP1743 the metapopulation is explicitly involved.
5: In Rb Leeuwarden 23 January 2004 LJN: AO2334 competition between various species of mice is mentioned.
6: In ABRvS 5 September 2003 uncertainty with regard to the effects of cockle fishing is involved.

Chapter VII

Synthesis and conclusions

Integration in river management

1 Introduction

1.1 Scope and goals of the thesis

The scope of this thesis is nature conservation in river management and, within this context, the relation between ecological knowledge and information and legal instruments. The international political and legal framework for nature conservation clearly states that conservation and rehabilitation of biodiversity, referring to the diversity of life on Earth, is its main goal. River-floodplain ecosystems have high biodiversity potential and therefore play an important role in nature conservation. In north-western Europe, rivers are characterised by severely impoverished biodiversity levels (Petts, 1989). Moreover, large scale reconstruction measures are currently being planned and carried out in the winter beds of these rivers (e.g. flood defence measures, ecological rehabilitation and hydraulic engineering works. This makes assessment of effects of river management on biodiversity a major issue.

Rotmans ***et al.*** (1996) define integrated assessment as an interdisciplinary process of combining, interpreting and communicating knowledge from different scientific disciplines, which provides useful information to decision makers. For consistent river management, which optimises the balance between nature conservation and other interests, assessment models are required that can integrate political-legal considerations with ecological-scientific knowledge concerning biodiversity. These models can:

1. Facilitate the input of ecological knowledge and information into decision making processes, and help gaining insight into an enormous complexity.
2. Give insight into the effects of river management on protected and endangered species.
3. Show the implications for river management of legal instruments for nature conservation, through valuation of biodiversity features based on their relevance to policy and legislation.
4. Contribute to reconstruction and management designs in which different river management goals (e.g. nature conservation, ecological rehabilitation and flood defence) are mutually attuned.

At the beginning of this study, models for linking political-legal considerations with ecological-scientific information for the purposes mentioned above were not available. In other words, there was a gap in the instrumentation for river management.

The goal of this thesis was to design, apply and evaluate a scientifically underpinned model for integration of ecological knowledge and information with legal instruments for nature conservation in river management, and to show its possibilities and limitations for application in evaluation studies and impact assessment. Furthermore, the study aimed to show the effects of river management and physical planning for protected and endangered biodiversity. Last but not least, I intended to provide useful insights into the relationship between ecological science and nature conservation legislation. This is necessary for finding opportunities for combination and mutual adaptation of ecological knowledge and legal instruments for nature conservation in river management. This combination and mutual adaptation is the definition of integration used in this thesis.

1.2 Goals of this synthesis

With the model BIO-SAFE (Spreadsheet Application For Evaluation of BIOdiversity), steps have been taken to fill the abovementioned gap in the instrumentation for river management. However, the possibilities and limitations of the model must be clear and placed in the broader context of river management practice, ecological theory and legal procedures.

The goal of this synthesis is to answer the research questions as formulated in chapter I. The model BIO-SAFE is evaluated with respect to its incorporation of ecological and legal aspects of nature conservation within the context of river management, and its complementarity, indicator function, validity and sensitivity. Consequences of river management measures for protected and endangered biodiversity are summarised. Conclusions and recommendations are given for optimisation of river management in terms of nature conservation, as well as for the mutual attuning of ecological knowledge and legal instruments for nature conservation within the context of river management.

Ecological and legal aspects concern ecological knowledge and information required for the scientific underpinning of nature conservation, and procedures given in legal instruments. The legal procedures used for model evaluation are derived from the Habitats Directive, Birds Directive, and the Dutch Flora and Fauna Act. The Water Framework Directive was not included because during the course of this study, the selection of species for reference conditions for various water types in European river basins was not yet finished. Therefore no species based model could be developed for this Directive. Moreover, this Directive predominantly focuses on water bodies and largely ignores semi-aquatic and terrestrial parts of river floodplain ecosystems.

Complementarity refers to useful extra information yielded by BIO-SAFE, when compared to a conventional ecological approach (in this case species richness). Indicator function refers to the correlation between BIO-SAFE output and species richness. Validity is defined as the correlation between effects on protected and endangered biodiversity predicted by BIO-SAFE based on ecotope data, and observed effects on protected and endangered biodiversity. Sensitivity is the influence of the value assignment in the model on the outcomes of ranking different reconstruction alternatives in impact assessments. The complementarity, indicator function, validity and sensitivity are evaluated only for the Dutch version of BIO-SAFE, because of lack of data concerning reconstructed and/or rehabilitated floodplains and flood defence scenarios in Germany, France and Belgium.

1.3 Outline

Paragraph 2.1 describes how ecological knowledge and the species approach of nature conservation legislation were combined in the model BIO-SAFE in a way that is suitable for river management (Figure 1). This description is based on chapter II. In paragraph 2.2, the model is also evaluated with respect to its possibilities for application in river management. Subsequently, the incorporation of ecological and legal aspects is reviewed using the results of chapter VI, which compared ecological science with legislation.

The statistical analysis of the results of application of BIO-SAFE in evaluation of floodplain measures and impact assessments (chapter III and IV) form the basis for evaluation of its complementarity, indicator function, validity, and sensitivity (Figure 1). In paragraph 3, the results of the application of BIO-SAFE in ecotope valuation and the evaluation of various river floodplain reconstruction measures for protected and endangered biodiversity are reviewed. This review is based on chapter II, III, IV and V, and used for recommendations for optimisation of river management. Paragraph 4 discusses the possibilities and limitations for integration of ecological knowledge with legal instruments for nature conservation, and places the results of this thesis within the broader context of integral river management. The overall conclusions and recommendations are given in paragraph 5.

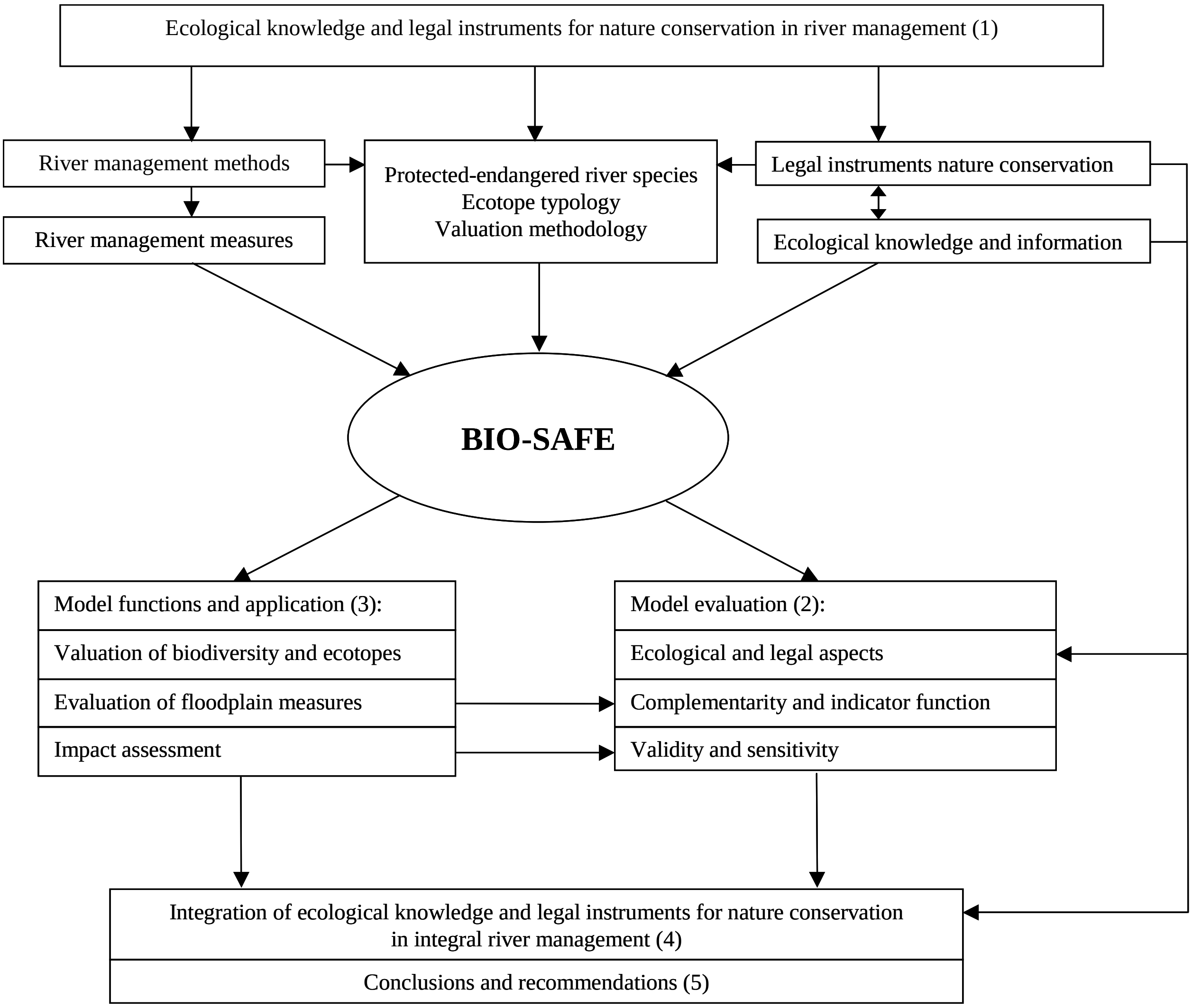

Figure 1. Flowchart of this synthesis, the numbers between brackets refer to the paragraphs.

2 Model evaluation

2.1 Model description

The first research question of this thesis (chapter I) asked how a transnational model (BIO-SAFE) could be developed for evaluation of and impact assessment for river-floodplain ecosystems, integrating legal and policy instruments for nature conservation with ecological knowledge into river management.

BIO-SAFE has been developed for the rivers Rhine and Meuse in the Netherlands, Germany, France and Belgium, based on protected and endangered species, and their habitats described by means of ecotopes.

The BIO-SAFE model has been built using three components: 1) a species selection, 2) an ecotope typology, and 3) a valuation methodology. Every component has been constructed using scientific methods, while incorporating methods and instruments of river managers and policy makers (Figure 1).

Species selection

The species in the model are river characteristic, protected and/or endangered species that are indigenous to Germany, France, Belgium and the Netherlands. Taxonomic groups involved are higher plants, birds, herpetofauna, mammals, fish, butterflies and dragon- and damselflies. Table 1 lists the numbers of species per country and taxonomic group and shows that many species are protected, but not red-listed. Differences between countries can be explained by biogeographical aspects and differences in environmental pressure on species in each country. In addition, sometimes the criteria for selection of red-listed species are different.

The term ‘protected and/or endangered’ is defined here as explicitly mentioned in the EU Birds and Habitats Directive, the Bern and Bonn conventions and/or Red Lists (selection criteria applicable in all four countries). This also includes species for which protected habitat areas must be designated. In addition, species protected by the Dutch Flora and Fauna Act were selected for a version of BIO-SAFE designed for a Strategic Environmental Assessment of the Key Decision on the spatial plan Room for the River (SEA-RfR) commissioned by the Dutch government (Dutch Ministry of Transport, Public Works and Water Management, 2005).

Table 1. Numbers of species meeting the selection criteria, per taxonomic group, per country. Between brackets: numbers of species that are protected, but not red-listed.

Taxon	End selection				
	NL	NL-FFA	G	F	B
Higher plants	136 (0)	140 (9)	60 (12)	12 (*)	90 (2)
Birds	60 (33)	73 (46)	58 (34)	113 (66)	38 (15)
Reptiles and Amphibians	9 (3)	7 (1)	11 (0)	7 (4)	4 (1)
Mammals	9 (3)	9 (3)	11 (0)	7 (4)	5 (1)
Fish	20 (1)	20 (1)	17 (1)	10 (1)	16 (0)
Butterflies	17 (0)	21 (0)	9 (2)	7 (0)	15 (0)
Dragon- and Damselflies	6 (0)	6 (0)	5 (5)	4 (2)	5 (0)
All groups	257 (40)	276 (60)	171 (54)	160 (77)	173 (19)

* Only regionally protected species were selected, which are also very rare.

The criterion ‘river characteristic’ means in this study that, according to the literature and a panel of experts, the whole population of a species or the largest part of the population is considered to be river-bound or closely associated with riverine areas. Species that are currently not found in riverine areas because sufficient habitat is lacking, but prefer river-floodplain ecosystems, were also considered characteristic of rivers.

Ecotope typology

Ecotopes are defined as spatial units of a certain extent, which are relatively homogeneous in terms of vegetation structure, succession stage and the main abiotic site factors that are relevant to plant growth (Klijn & Udo de Haes, 1994). River ecotopes are identified on the scale of the riverine landscape, on the basis of hydrodynamics, morphodynamics, management dynamics and land use (Van der Molen ***et al.,*** 2003). Floodplain reconstruction planners frequently use ecotopes for model calculations and the drawing up of reconstruction plans. Therefore, the concept of ecotopes can be used as a common language for hydrologists, ecologists and landscape designers.

The typology was constructed using typologies that are applicable to river floodplain ecosystems in a transnational context (i.e. the Netherlands, Germany, Belgium, and France), and cover four different levels of spatial scale: 1) the CORINE Land Cover classification (1:100,000), 2) the typology of the International Commission for the Protection of the Rhine

(1:50,000) and 3) the Dutch River Ecotope System (1:25,000 and 1:10,000). The ecotope typologies are all based on hydrodynamics as well as vegetation structure except for the CORINE Land Cover classification which is based on vegetation structure only. The different levels of spatial scale offer the opportunity for up- and downscaling of model output and enable BIO-SAFE to process input data on various levels of scale.

Valuation methodology
Values were assigned to protected and endangered species, quantifying their status according to various legal instruments and the relative importance of these instruments, i.e. the European Habitats Directive, the European Birds Directive, the Conventions of Bern and Bonn and Red Lists. These instruments (valuation criteria) were assigned different weights by a panel of experts, based on the experts' perception of the relevance of the criteria to river management and policy. In a version of BIO-SAFE designed for the SEA-RfR, weights were assigned to species based on their legal status according to the Dutch Flora and Fauna Act. Through the linkage of species to ecotopes, values are assigned to ecotopes as well. Although subjective, valuation based on legal and policy status is often applied in policy, research and model development (Freitag ***et al.***, 1997; Oertli ***et al.***, 2002; Ten Brink ***et al.***, 2001).

2.2 Suitability for river management

A primary aim of this thesis was to develop an operational model for integration of ecological knowledge with legal instruments in river management. BIO-SAFE can assess actual and potential values of river floodplains and ecotopes in these areas, and reconstruction plans based on 257 protected and endangered river characteristic species and their habitats (Figure 1). BIO-SAFE can therefore be used as a tool for biodiversity assessment with regard to design and evaluation of physical planning projects, management measures, Environmental Impact Assessments, and comparative landscape-ecological studies.

BIO-SAFE is well adapted to the methods of river management in north-western Europe, that characterise river-floodplain ecosystems by means of ecotopes. Moreover, the model can already be used when only information on presence/absence of species and ecotopes is available. This makes the model of high practical value at initial phases of planning processes when much data is lacking (e.g. physical and chemical conditions, and the abundance of species). Moreover, legal procedures require first of all presence/absence information on protected species.

The usefulness of the model was shown in a large number of case studies, within the context of the EU IRMA-SPONGE project (De Nooij ***et al.***, 2001; Klijn ***et al.***, 2004), and within the context of the SEA-RfR for reconstruction of the Dutch floodplains of the river Rhine, aimed at flood defence and ecological rehabilitation (Dutch Ministry of Transport, Public Works and Water Management, 2005). There are, however, a number of ecological and legal aspects of the model that require closer scrutiny. Research question 7 (chapter I) asked to what extent BIO-SAFE integrates ecological and legal aspects of nature conservation. The next two paragraphs answer this question.

2.3 Ecological aspects of nature conservation in river management

Ecological aspects used for evaluation of BIO-SAFE are the three levels of biodiversity (genetic, species and ecosystem; UNEP, 1992), the three components of biodiversity (composition, structure in space and time, and processes; Noss, 1990), and life history traits of species.

Composition refers to presence and abundance of flora and fauna species and populations, types of ecosystems in the area and local genetic varieties. Quantification of composition can include information on numbers of individuals and numbers of species per habitat patch or taxonomic group (abundance, evenness; e.g. the Shannon-index and Menhinnick's index; Magurran, 1988). Structure describes how the elements of biodiversity, including genes, habitats, geomorphic patterns and cyclic phenomena are organised in space and time (i.e. spatial and temporal patterns). Functional diversity refers to physical, biological or bio-physical processes structuring ecosystems and communities, such as disturbance regimes, succession, population dynamics, life history and gene flow (Ward ***et al.***, 1999). The life history of a species includes habitat use, migration behaviour, interaction with other species, its tolerance for environmental factors, its response to stress and its (meta)population dynamics.

Ecological aspects in BIO-SAFE

BIO-SAFE concerns the species and ecosystem level of biodiversity, quantified based on presence of species and ecotopes. With respect to compositional diversity, it should be mentioned that BIO-SAFE incorporates protected and endangered species that are characteristic of river systems, which has a very strong ecological significance. Because the species in the model are for a large part dependent on riverine habitats in floodplains, they are the species that suffer the worst from negative impacts and benefit the most from positive effects of floodplain reconstruction.

The structure component of biodiversity in space and time is integrated in the model BIO-SAFE by means of ecotopes. Landscape ecological classification is an essential component of landscape ecological studies aimed at valuation of areas. It is the starting point for analysis and understanding, which requires definition of units within a complex system (Klijn & Udo de Haes, 1994). As mentioned in paragraph 1.2, ecotopes were used because it is the methodology of river managers for planning. The typology is the basis for linking species to landscape ecological characteristics of river-floodplain systems.

This approach to composition and structure enables the model to use input data on presence of species and/or surface area of ecotopes on four levels of spatial scale for:

1. Valuation of actual situations on the level of ecotopes and areas, reflecting the importance of an ecotope or area for a species group (e.g. birds) and the degree to which the maximum potential value for a species group has been achieved in an actual situation.
2. Valuation of potential situations on the level of ecotopes and areas, reflecting the importance of an ecotope, area or a scenario/reconstruction alternative for a species group.
3. Trend analysis (retro- and prospective) reflecting dynamics of the importance of an area for the different species groups.
4. Valuation of hydrodynamics, morphodynamics, vegetation structure, land-use and management, on the basis of the potential value of ecotopes for different species groups.

Many processes can be of importance for creation and maintenance of composition and structure. A key process is defined as a process that plays a dominant role in structuring or maintaining ecological units (population, habitat, community, ecosystem, and landscape) and/or in structuring or maintaining processes between units. Key processes may be of a completely abiotic nature, biotic nature, or a mix of both. In the model BIO-SAFE, key processes of river systems (hydro- and morphodynamic processes) form the basis for the ecotope system (Van der Molen ***et al.***, 2003). Life history traits of species in BIO-SAFE concern habitat use, which is a fundamental aspect of species life history.

Limitations

Concerning composition, the model does not account for the genetic level of biodiversity. Furthermore, species that are not protected and/or endangered river characteristic species, are left out. Many protected species are not the species with the highest extinction risk or importance for ecosystems (chapter VI; Van der Velde ***et al.***, 1994). Table 1 clearly shows that more than half of the protected river characteristic birds in the model are not endangered according to Red List criteria. For mammals and herpetofauna, this proportion is one third. In any case, vascular plants and vertebrates together make up less than 10% of known biodiversity (Franklin, 1993). Therefore, a particular set of species is an extremely small fraction of the biota in any river floodplain. However, a relatively small set of species can already give a lot of information about the ecological quality of an area (Brooker, 2002).

Because the selection of riverine species is partly based on expert judgement, it carries a subjective touch. However, there seems to be no way to renounce this subjectivity because there are no data available that allow a selection based on mathematical or logical algorithms (Buijse ***et al.***, 2005).

The fact that BIO-SAFE uses only presence/absence information on species means that one individual of a species already makes an area valuable in terms of that species. Valuation of actual situations, or evaluation of trend data, may therefore over- or underestimate the actual importance or development of the biodiversity in the area.

Structural diversity in space and time is reduced to the presence and abundance of ecotopes. This accounts only for one aspect of the temporal pattern of hydrodynamics (number of days flooded per year), the intensity of morphodynamics (centimetres erosion/sedimentation per year), and intensity of land use (Van der Molen ***et al.***, 2003). Topological relationships, shape of ecotopes, and similar aspects of spatial structure (Farina, 1998), and aspects of temporal structure such as inundation duration, frequency, water level fluctuations and timing of inundations (however ecologically relevant) are not taken into account. Habitat suitability analyses should consider species' demands concerning surface area, connectivity, heterogeneity and configuration of habitat patches (Southwood, 1977; Verboom ***et al.***, 2001). Also chemical aspects are important. BIO-SAFE does not take these aspects of habitat suitability into account. Ecotopes are a simplification of a species' habitat, required for modelling activities, visualisation and input-output relations with other models. The model is therefore prone to overestimating the potential value of an ecotope situation.

Many processes (on the level of ecosystems, species-populations and genes) are not included in BIO-SAFE. Linking species to ecotopes can not account for all physical, chemical and biological causal factors and processes that determine the potential value of a landscape to species and populations. Life history traits such as migration behaviour, interaction with other species, tolerance for environmental factors, response to stress and (meta)population dynamics are impossible to model for 257 species, because the required data is not available for most species.

Improvement of BIO-SAFE

The model can be adapted in such a way that it can use data on species abundance as well. More attention can be paid to ecological characteristics of selected species, by using a more detailed ecotope classification. This has already been done for higher plants, using a typology that incorporates moisture regime, acidity and nutrient levels (Runhaar ***et al.***, 1987). Furthermore, the model can easily be adapted in order to account for the fact that some ecotopes are more important than others as habitat, and that combinations of ecotopes are required, some ecotopes being complementary (both required) and others supplementary. The model can also be optimised by setting minimum required ecotope surface area thresholds, required for underpinning the estimation of potentials (Eiswerth & Haney, 2001;

Mac Nelly ***et al.***, 2003; Huggett, 2005). The concept of Minimum Area Requirements for a Minimum Viable Population (MAR, and MVP; Soulé, 1987; Verboom ***et al.***, 2001) can be integrated in BIO-SAFE when this information will be available for the species in the model. Incorporation of algorithms that can describe non-linear relationships between ecotope surface areas and biodiversity values for species groups (Oertli ***et al.***, 2002) is also recommended. These relations can be derived from case studies and existing literature. Information on recovery capacity of species by migration and / or reproduction might be used in the future to account for species' vulnerability to impacts.

2.4 Legal aspects of nature conservation in river management

Legal aspects used to evaluate BIO-SAFE concern the information required in legal procedures of the Dutch Flora and Fauna Act (FFA), the EU Birds Directive (BD, Council Directive 79/409/EEC) and EU Habitats Directive (HD, Council Directive 92/43/EEC). The HD and BD give regulations for species protection and protection of areas. Protected areas are designated (HD art. 3, BD art. 3) based on the distribution and requirements of protected species as well as the so called 'Natural Habitats': terrestrial or aquatic areas distinguished by geographic, abiotic and biotic features, whether entirely natural or semi-natural.

Legal procedures concern protected species and protected areas (Special Areas of Conservation, SACs), and have different information needs. Species protection requires information regarding:

1. Presence of protected species and their legal status according to the FFA (this includes species protected by the HD art. 12-15, annex IV and V; BD art. 1; Convention of Bern art. 5 and 6, annex I and II; Convention of Bonn art. 3 and 4, annex I and II).
2. Habitat of protected species and the disturbance of this habitat resulting from human activity (FFA art. 8 & 11).
3. Stressing of individuals of protected species caused by human activity (FFA art. 10).
4. Effects on the populations of protected species caused by human activity (FFA art. 75) and the extinction risk (conservation status) of species concerned.
5. Presence of protected habitat and presence of species for which an area or landscape element was designated (FFA art. 19).

Area protection requires information regarding:

1. Presence of SACs.
2. Presence of species and natural habitats for which the SAC was designated (these species and natural habitats are given in HD annex I and II, and BD annex II).
3. The qualifying species' habitats and disturbance of habitat caused by human activity, function of the SAC for these species and the structure and dynamics of the SAC (HD art. 6).
4. Effects on the populations of qualifying species caused by human activity (HD art. 6), extinction risk (conservation status) of species concerned, and the effects on the coherence of Natura 2000.
5. Measures necessary for mitigation and/or compensation of the effects of human activity on the qualifying species (HD art. 6). For each affected species the required habitat acreage, habitat quality (physical, chemical, biological factors), habitat connectivity and optimal location of new habitat, necessary for maintaining the coherence of metapopulation must be known.

Possibilities for application in legal procedures
BIO-SAFE yields information regarding the degree to which various river management measures or actual situations meet goals and criteria set in nature conservation legislation. It also translates legal obligations regarding species into information about which ecotopes in a floodplain area are actually or potentially most valuable. BIO-SAFE can therefore be used as an early-warning system in legal procedures.

The species selection in BIO-SAFE includes bird species mentioned in Annex I of the BD, species mentioned in Annexes II, IV and V of the HD, species mentioned in Appendices I or II of the Convention of Bonn (CBo, Intergovernmental Treaty, Bonn 1.XI.1983), species mentioned in Appendices I or II of the Convention of Bern (CBe, Council of Europe, Bern 19.IX.1979, European Treaty Series/104). BIO-SAFE also includes river characteristic species protected by the FFA. Furthermore, species meeting national Red Data list criteria according to the World Conservation Union (IUCN, 1993, 1994, 2001) are incorporated.

In legal procedures concerning species protection, BIO-SAFE can give information on the presence, habitat demands, and extinction risk of the river species protected by the legal instruments mentioned in the previous paragraph. In procedures that deal with protected areas BIO-SAFE can give information on the presence, habitat, and extinction risk of river species for which the area was designated and can help determine potential value of compensation measures.

Limitations
BIO-SAFE is useful only in parts of the legal procedures given above. For species protection, the model cannot determine stressing of individuals, population effects, and presence of protected habitat and landscape elements. For area protection, presence of SACs, presence of natural habitats, effects on (meta)populations, and the coherence of Natura 2000 can not be assessed. Furthermore, the information on the habitat provided by BIO-SAFE is only limited, as discussed in paragraph 2.1. Because of these limitations BIO-SAFE is best suitable for relatively coarse level effect studies, early in the planning process. The detailed information required by legal procedures must be derived from field-studies, autecological data, population data and empirical cause-effect studies.

BIO-SAFE was constructed for Rhine and Meuse specifically and is not applicable outside the winter bed area of the major channels within the catchments of these two rivers.

The species selection of the model presented in this thesis did not include all the river characteristic species protected by the FFA. However, a version of BIO-SAFE which does include these species has already been constructed. Valuation of ecotopes does not use the natural habitats defined in Annex I of the Habitats Directive (Natura 2000).

There are a number of aspects of BIO-SAFE that do not entirely correspond with legislation:

1. Only river characteristic species are included, whereas legislation requires that all protected species are taken into account.
2. Also endangered species (Red List criterion) are included, whereas for legislation only legally protected species are required. However, the Red List criterion can be switched off in the model.
3. BIO-SAFE assesses effects on the level of taxonomic groups or all taxonomic groups taken together, whereas assessments within legal procedures must be done for each separate species. The BIO-SAFE approach leads to trading off one species or one taxonomic group against another, which does not comply with legal regulations.
4. The assignment of values to species (that sums up weights given to international legal instruments and Red Lists) differs from a legal approach, which does not quantify legal status and does not allow summation of different legal instruments. Within the

legal framework a difference is made in qualitative terms like 'strictly protected' and 'less strictly protected', and the Habitat Directive ***substitutes*** the Conventions of Bern and Bonn rather than being an addition.

5. Species protection is regulated by the Flora and Fauna Act. The version of BIO-SAFE mostly applied in this thesis is based on the Habitat Directive, the Birds Directive and the Conventions of Bern and Bonn. However, a version of BIO-SAFE based on the FFA was also developed.

Improvement of BIO-SAFE

Future development and adaptation of BIO-SAFE can extend the possibilities for application in legal procedures. For example, it can be adapted to other ecosystems in order to be applicable outside river floodplains, and the 'natural habitats' of the Habitats Directive can be incorporated. Incorporation of the selection of SACs into the model will make BIO-SAFE applicable within the context of area protection as well.

In order to increase applicability in legal procedures, BIO-SAFE must be made suitable for impact assessments on the level of species and not only on the level of taxonomic groups. The valuation methodology of BIO-SAFE can be made more suitable within a legal context by assigning values to species in a way that better accounts for the legal status of protected species. For example, the fact that the Habitats Directive is considered to be a substitution of the Convention of Bern, rather than an addition, can be accounted for.

2.5 Indicator function and complementarity

The second research question was: To what extent does BIO-SAFE yield information indicative for or complementary to a conventional biodiversity quantification method?

Indicator function

Indicator function is defined here as the positive correlation between BIO-SAFE output and species richness. Chapter III evaluated the effects of ecological rehabilitation on biodiversity of higher plants, birds, mammals, herpetofauna, odonates and butterflies in floodplains along lowland rivers in the Netherlands, using two different approaches.

The Analysis of Covariance in chapter III showed that there is a correlation between species richness and BIO-SAFE output in all sites for birds, herpetofauna, odonates and, to a lesser degree, higher plants. The patterns of potentials found for protected and endangered biodiversity along a hydrodynamical gradient (chapter V) match very well with patterns of species richness along similar gradients (Van den Brink, 1994; Van den Brink ***et al.***, 1996; Ward & Tockner, 2001; Aarts ***et al.***, 2004; Chovanec ***et al.***, 2005). In paragraph 3.3 these patterns are given. The congruence between our results and the results of these other studies is evidence for an indicator function of BIO-SAFE output on the system level.

Complementarity

Complementarity refers to useful extra information yielded by policy and legislation based assessment, when compared to species richness. For the groups higher plants, herpetofauna and odonates, the model gives complementary information when applied to evaluation of rehabilitation measures (chapter III). If species richness increases but the BIO-SAFE index does not, then the point can be made that measures taken as such are adequate, but need to be optimised in order to enhance possibilities for protected and endangered river species. Conversely, when an evaluation indicates that neither species richness nor a policy based index increases, then the recommendation to reconsider rehabilitation has a much stronger foundation.

2.6 Validity and sensitivity

Research question 3 was: How valid and sensitive is assessment of impacts of floodplain reconstruction using BIO-SAFE?

Validity

Validity is defined here as the correlation between effects on protected and endangered biodiversity predicted by BIO-SAFE based on ecotope data, and observed effects on protected and endangered biodiversity. In chapter IV the validity of the model has been tested using data on five reconstructed floodplains of the river Rhine. Changes in the presence and surface area of ecotopes were used to predict changes in protected and endangered biodiversity values. A statistically significant and positive correlation between predicted and observed values for protected and endangered species was found. Based on these results we conclude that BIO-SAFE can adequately predict impact on protected and endangered species for the five study areas. This is a strong indication that BIO-SAFE can also give adequate impact assessments in other river floodplain ecosystems in the Netherlands.

The validity test had a limited character. The dataset used covered five areas and a time span of less than five years, and it was only done for the river Rhine in the Netherlands, not for the river Meuse. A more comprehensive and detailed validation study should be carried out to prove the model's validity more rigorously. Moreover, it should be investigated for which species groups BIO-SAFE predicts well, and for which groups it doesn't. It can then be seen if the model needs extra parameters to enhance the validity of impact assessment.

Sensitivity

Assessments using BIO-SAFE presented in this thesis have all used a value assignment given by a panel of specialists composed of scientists, policy makers and conservationists. Because value assignment always remains somewhat subjective, the question arose whether value assignment has a large influence on ranking different reconstruction alternatives, leading to uncertainty in impact assessment (cf. Geneletti ***et al.***, 2003).

The sensitivity of BIO-SAFE to value assignment has been analysed in chapter IV by applying the model to assess impacts of physical reconstruction alternatives for Dutch floodplains, aimed at flood defence and ecological rehabilitation. The weights given to different valuation criteria for protected and endangered species (i.e. EU Birds and Habitats Directive, Conventions of Bern and Bonn and Red Lists), were varied randomly and the effects on ranking of alternatives were analysed.

Indices for the taxonomic groups birds, herpetofauna, dragon- & damselflies and for all taxonomic groups combined were not sensitive at all to value assignment. Indices for higher plants, butterflies and mammals were only slightly sensitive to the weight given to the Red List criterion. Indices for mammals and butterflies are also sensitive to the Habitats Directive criterion. It proved that as long as all criteria were taken into account there were no significant shifts in ranking orders of reconstruction alternatives.

It is concluded that the value assignment does not lead to significant uncertainty in impact assessment with BIO-SAFE. However, the decision to include valuation criteria or not is very important for assessment outcomes. Therefore, the model is best used with all criteria included in the assessment.

3 Model application

In the previous paragraphs it was shown that BIO-SAFE yields meaningful information and is valid enough for drawing conclusions about river management in relation to nature conservation legislation. In this part of the synthesis, the results of the application of BIO-SAFE in ecotope valuation (3.1), evaluation of floodplain measures (3.2) and impact assessment – scenario analysis (3.3) are reviewed. The results are used to discuss the consequences of river management measures for protected and endangered biodiversity, and to give recommendations for optimisation of river management (research questions 4 & 5 of Chapter I).

3.1 Ecotope valuation

Ecotopes and vegetation structure

In chapter II, the importance of river ecotopes for protected and endangered species was studied. Table 2 presents an example of ecotope valuation. Ecotope importance is strongly determined by the spatial and biological level and the policy and legislation criteria that are used for valuation. However, general statements about the implications of habitat demands of protected and endangered species can be made.

Table 2. TEI constants for the Netherlands, reflecting the importance of ecotopes of level 3 (1:25,000) and using all valuation criteria.

Ecotopes		TEI (Taxonomic group Ecotope Importance)						
		HP	BI	HF	MA	FI	BU	DD
Sd	Deep summer bed	0	13	0	28	73	0	0
Ss	Shallow summer bed	1	10	10	44	85	0	61
Ws	Side channel	1	25	15	69	45	0	38
Sb	Beach, Bank, Bar	14	13	0	56	47	0	69
Wf	Floodplain channel	3	64	100	69	44	0	31
Wl	Lake	1	40	100	69	27	0	0
Mh	Herbaceous marsh	19	42	76	59	3	5	23
D1 Lh-1	River dune	17	0	44	0	0	9	0
D1 Lh-5	Gravel deposit	1	0	0	0	0	14	0
Mg	Marsh grassland	5	32	85	34	0	36	61
Fg	Moist grassland	11	33	40	34	0	36	61
Fh	Herbaceous moist floodplain	5	1	42	18	0	32	61
Lg	Levee pasture	43	8	44	17	0	32	0
Lh	Herbaceous levee or dyke	18	8	59	5	0	5	61
Hg	High-water-free grassland	28	7	0	21	0	32	61
Hh	High-water-free herbaceous area	6	7	59	5	0	5	61
T1-F	Shrubs in floodplain	1	9	52	35	0	9	69
T1-L	Shrubs on levee	4	1	67	18	0	18	69
Lf	Forested levee	5	8	67	46	0	23	8
Hf	High-water-free forested area	3	5	67	46	0	14	69
B2.3	Softwood alluvial forest	1	11	52	35	0	14	8
B3.3	Hardwood alluvial forest	2	5	67	46	0	14	8
B4.3	Other characteristic forested biotopes in floodplains	0	5	52	30	0	0	8

HP: Higher Plants, BI: Birds, HF: Herpetofauna, MA: Mammals, FI: Fish, BU: Butterflies, DD: Dragon- and Damselflies, In Table 2, it can be seen that every taxonomic group is, through their habitat demands, related to different sets of ecotopes. The most valuable ecotopes for higher plants are pioneer ecotopes such as beaches, banks, bars and river dunes. Furthermore, terrestrialising ecotopes like herbaceous march and moist grassland, and high water free ecotopes like levee pastures, high-water free grassland and herbaceous levees are important to higher plants. Birds favour aquatic ecotopes like side channels, floodplain channels and lakes, as well as herbaceous marsh, marsh grassland and moist grassland. For mammals the pattern is largely similar. Mammals, however, also favour forested ecotopes. Also herpetofauna have a pattern similar to birds but for this group, also shrubby vegetations are important. Protected and endangered fish largely live in the shallow and deep summerbed, but also side channels and river bank ecotopes are important. For butterflies marsh and moist floodplain grassland, herbaceous moist floodplain and dry wood-free biotopes like levee pastures and high water free herbaceous areas are most important. These ecotopes are also favoured by dragon- and damselflies. This last group also favours floodplain channels and shrubs/woodland fringes and dry forested ecotopes.

Ecotopes and the hydrodynamic gradient

Chapter V examined the importance of different hydrodynamic conditions to protected and endangered riverine species. Importance is defined as the potential habitat function of a hydrodynamic situation for these species.

When all taxonomic groups of protected and endangered species are combined, the biodiversity potential shows an increase with decreasing hydrodynamics, until class 6, where an optimum is reached (Figure 2). In this class the highest number of protected and endangered species occurs. The contribution of the different taxonomic groups to the potential differs markedly along the hydrodynamic gradient. Many species of higher plants, fish and butterflies have a narrow range for hydrodynamics (data shown in chapter V).

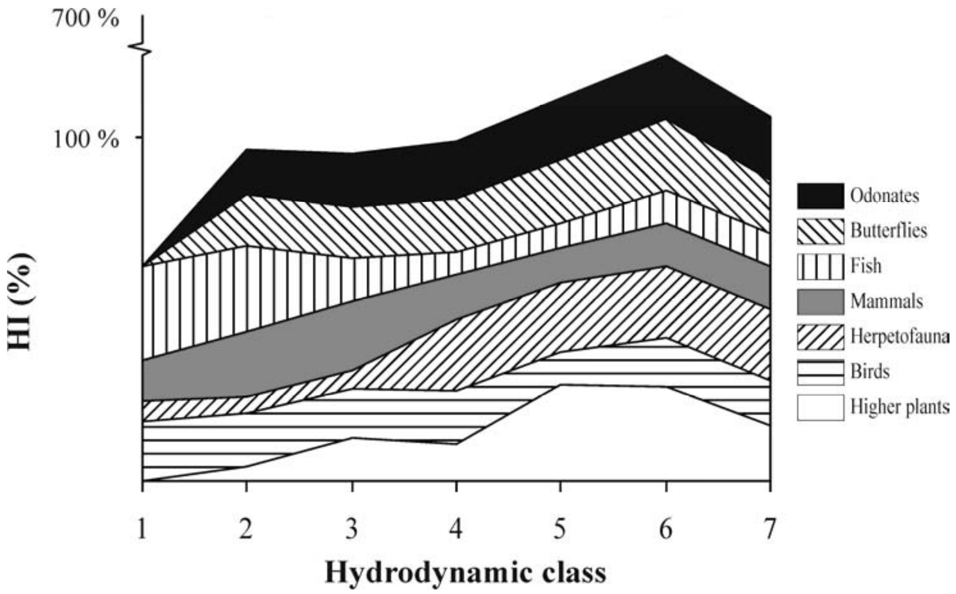

Figure 2. Importance of hydrodynamic conditions (Hydroclass Importance, HI) for all species groups combined, along the hydrodynamic gradient. 1: highest degree of hydrodynamics (e.g. main channel); 7: lowest degree of hydrodynamics (e.g. natural levees).

High dynamic conditions are favoured by fish, intermediate and lower dynamic situations are most important for higher plants, herpetofauna and butterflies. Birds, mammals and dragonand damselflies (odonates), are found along the entire hydrodynamic gradient. The results show that the entire hydrodynamic gradient is important. The different taxonomic groups have different relationships with hydrodynamics (in terms of days flooded per year) in such a way that every part of the gradient provides optimal potential for one or more taxonomic groups. A remarkable result is the importance of low-dynamic parts, which are most important for the groups of higher plants, butterflies, herpetofauna and odonates, and give the highest overall potential.

Optimisation of ecotope situations for protected and endangered biodiversity
Because different sets of ecotopes are most important to different taxonomic groups, and every part of the hydrodynamic gradient has optimal potential for one or more taxonomic groups, it is concluded that:

1. River reconstruction and management should aim at re-establishing the entire hydrodynamic gradient, maximising landscape heterogeneity.
2. Lowering floodplains is best combined with measures that enlarge the flooding area, resulting in floodplains where dry as well as wet ecotopes are represented ideally along with an intact gradient from wet to dry.
3. When this is not possible, spatial and temporal heterogeneity of hydrodynamic conditions should be maximised within the spatial limits.

3.2 Evaluation of measures taken in river-floodplain

In this paragraph measures taken in floodplains in the Netherlands are evaluated using BIO-SAFE. In total, biodiversity trends of fifteen floodplains that were reconstructed or rehabilitated in the past have been studied in this thesis. Ten floodplains were rehabilitated only by means of a change of management; five floodplains were also partly excavated. The measures included a management change from agricultural management to a strategy that includes low-density grazing by horses and cattle, and increased influence of river dynamics in the floodplain (clay excavation, the reopening of secondary channels, removal of summer dikes) (Pelsma, 2002; Aggenbach & Pelsma, 2005).

Change of management
In chapter III it was shown that for most taxonomic groups, biodiversity in ten rehabilitated floodplains significantly increased within approximately seven years after initiation of rehabilitation. However, the outcome of the evaluation differs per taxonomic group and approach (Table 3). The results show positive temporal trends with respect to species richness of higher plants, herpetofauna and odonates, and to a lesser degree mammals and butterflies. Values for protected and endangered species (TBS) increased to some extent, but with a lower level of significance, for higher plants, herpetofauna and butterflies.

Table 3. Trends of species richness (R) and values for protected and endangered river species (TBS), calculated based on temporal trends of ten floodplains in the Netherlands.

Approach	HP	BI	HF	MA	BU	DD
R	++	0	++	+	+	++
TBS	+	0	+	0	+	0

HP: Higher Plants, BI: Birds, HF: Herpetofauna, MA: Mammals, BU: Butterflies, DD: Dragon- and Damselflies
++: significant increase, $p < 0.0083$ (cf. Bonferroni-correction)
+: significant increase, $p < 0.05$
0: no significant increase

Excavation and change of management

Chapter IV gives results concerning observed effects of floodplain reconstruction on values for protected and endangered river species in four floodplains, with a recovery period of five years (Table 4; D ata sources: chapter IV). Only four reconstruction projects are given because the data for one of the areas was insufficient for drawing conclusions about reconstruction measures. Overall, there are strongly positive as well as strongly negative effects. Most negative effects occur in the 'Afferdensche en Deestsche Waarden' floodplain for the groups herpetofauna and mammals.

Table 4. Qualifications of observed effects of river floodplain reconstruction, measured with TBS (Taxonomic group Biodiversity Saturation).

Taxon	Ewijkse Plaat	Afferdensche en Deestsche Waarden	Blauwe Kamer	Duursche Waarden
HP	-	n.a.	n.a.	+
BI	0	+	++	+
HF	++	--	+	+
MA	+	--	0	++
BU	+	0	+	+
DD	0	++	++	+

HP: Higher Plants, BI: Birds, HF: Herpetofauna, MA: Mammals, BU: Butterflies, DD: Dragon- and Damselflies
+++: ≥ 40% increase, ++: ≥ 15%, < 40% increase, +: ≥ 2%, < 15% increase, 0: ≥ 0%, < 2% increase
-: > 0%, ≤15% decrease, --: > 15%, ≤ 40% decrease, ---: > 40% decrease of TBS indices relative to initial situation
n.a.: data not available.

Optimisation of rehabilitation and reconstruction measures

Based on Table 3 and Table 4, which concern rehabilitation and reconstruction of fourteen floodplains, it is concluded that the effects are promising but limited. Limited, because, rehabilitation did not have significant positive effects for protected and endangered river characteristic birds, mammals and odonates. The effects on higher plants, herpetofauna and butterflies, calculated by BIO-SAFE were less significant than effects on species richness of these groups. Reconstruction measures not only lead to positive effects, but sometimes also to highly negative effects ('Afferdensche en Deestsche Waarden' floodplains). These results might be due to the limited time span of the dataset used, i.e. a short recovery period, but reconstruction certainly has very large impacts on protected and endangered species.

The rehabilitation measures taken can be optimised in order to enhance possibilities for protected and endangered river species. Recent studies that evaluated effects of rehabilitation of Dutch floodplains (Nienhuis ***et al.*** 2002; Van der Molen & Buijse, 2005; Leuven ***et al.*** 2005) yield similar results. These studies conclude that rehabilitation measures must be carried out on a much larger scale in order to make significant contribution to rehabilitation of the river landscape in the Netherlands within safety boundary conditions. Furthermore, improvement of the chemical quality of water and soil must continue. Reconstruction designs will have to be conceived in a more sophisticated way in order to optimise opportunities for protected and endangered species, and to prevent problems with legal instruments due to important negative effects on protected species.

The results presented here can not be linked directly to different types of measures. This would require more detailed data about which measures led to which landscape changes, and data on more areas and longer time trends.

3.3 Impact assessment – scenario analysis

Many scenarios for floodplain reconstruction aimed at ecological rehabilitation and flood defence were assessed according to their effect on protected and endangered species (Table 5). All scenarios were realistic and existing scenarios planned by river authorities (data sources and locations: chapter II, IV and SEA-RfR).

Table 5. Qualification of effects of scenarios studied for floodplains of rivers Rhine and Meuse in the Netherlands. The alternatives with the best outcome for protected and endangered species are marked with an asterisk.

	Rijnwaarden				Common Meuse			IJssel2			Rijn5		
Taxon	R1	R2*	R3	R4	C1	C2	C3*	A1	A2	A3*	A1	A2	A3*
HP	+++	+++	++++	++++	-	---	---	-	-	+	-	-	0
BI	++	+++	+	+	0	++	++	0	+	+	-	-	0
HF	++	++	++	++	--	++++	++++	0	0	+	-	-	0
MA	++	++	0	0	0	0	++	0	0	+	-	-	0
FI	+	++	+	++	0	++	--	-	+	-	-	-	0
BU	++++	++++	+++	+++	-	---	---	-	-	++	0	0	0
DD	++++	++++	+++	+++	0	+	++	-	+	++	-	-	0
All	+++	+++	++	++	-	-	+	-	0	+	-	-	0

HP: Higher Plants, BI: Birds, HF: Herpetofauna, MA: Mammals, FI: Fish, BU: Butterflies, DD: Dragon- and Damselflies,
++++: ≥ 100% increase, +++: ≥ 40% < 100% increase, ++: ≥ 15%, < 40% increase, +: ≥ 2%, < 15% increase, 0: ≥ 0%, < 2% increase,
-: > 0%, ≤15% decrease, --: > 15%, ≤ 40% decrease, ---: > 40% decrease of scores relative to autonomous development
R1: Low Dynamics Scenario for a 16,000 $m^3 s^{-1}$ design discharge at Lobith; R2: Low Dynamics Scenario for a 18,000 $m^3 s^{-1}$ design discharge at Lobith; R3: High Dynamics Scenario for a 16,000 $m^3 s^{-1}$ design discharge at Lobith; R4: High Dynamics Scenario for a 18,000 $m^3 s^{-1}$ design discharge at Lobith.
C1: Agricultural land maintained as in a present situation and conservation of present natural values; C2: Development of nature and a decrease of agriculture; C3: Maximisation of natural processes.
A1: Focussed on keeping the costs within budget, relatively conservative; A2: Focussed on spatial quality, relatively nature oriented; A3: Preferred alternative, a combination of A1 and A2.

The results concerning Rijnwaarden show that the potentials for higher plants and insects selected for BIO-SAFE strongly increase in all reconstruction scenarios (Table 5). However, the scenarios aimed at low influence of river dynamics (R1 and R2) in the floodplain offer the best opportunities for all groups except plants. As far as birds are concerned, it may be noticed that especially the high dynamics scenarios (R3 and R4) result in considerably lower potentials than the low dynamics scenarios. Therefore, the scenario with the highest overall results is R2 (low dynamic scenario with high discharge capacity).

Scenario analysis for the Common Meuse (also known as Border Meuse) shows that the two scenarios that aim at development of new ecotopes (C2 and C3) are positive for most groups, especially for herpetofauna. However, the effects on plants and butterflies are so negative that only the most natural scenario (C3) has positive overall effects.

Concerning IJssel 2 and Rhine 5, it can be seen that A3 gives the best results for most groups. The most important characteristic of A3 is the large amount of natural grassland and natural levee pasture. Furthermore, this alternative is a compromise between a relatively conservative and a relatively nature development oriented option for the future.

Optimisation of reconstruction scenarios
Scenario analysis shows that the most natural situation, as envisaged by planners, does not necessarily correspond with the highest potential for protected and endangered river characteristic species. This can be explained by the importance of low dynamic ecotopes for higher plants, herpetofauna, butterflies and birds (meadow birds and geese). Measures aimed at flood defence, which include lowering of floodplains and river dike diversion (winter bed enlargement), may provide opportunities as well as threats for (protected) biodiversity.

Reconstruction designs should therefore aim at an optimal balance between high, intermediate and low dynamic conditions in river floodplain ecosystems, while creating a maximum of spatial heterogeneity (diversity of ecotopes). Our results show that scenarios that combine nature development and flood risk reduction can provide the best potentials for species mentioned in legal instruments. Therefore, the various goals of river management can be attuned in reconstruction designs. This will require tailor-made designs that provide extra space for low-dynamic ecotopes which can be created by over-dimensioning of flood defence measures such as river dike re-positioning, lowering of floodplains and widening or digging of secondary channels.

4 Ecological knowledge and legal instruments in integral river management

4.1 Integrating ecological knowledge with legal instruments in river management

The government develops policy on the basis of legal authority and carries out this policy with legal instruments. When nature conservation policy and legislation is concerned, ecology is an important science to provide the necessary knowledge and information. In chapter VI, the legal framework for nature conservation was compared with the ecological-scientific framework. This resulted in a number of conclusions concerning the relation between these frameworks in general, as well as for river floodplain ecosystems in particular.

In this paragraph differences between ecology and legislation, and the consequences of these differences for river management, are briefly summarised. Possibilities for integration of the two frameworks are given and related to various components of the legal framework, ecological research and model development. This paragraph answers research question 6 of Chapter I.

Differences between the ecological and legal framework
Chapter VI showed that the ecological and the legal framework differ with respect to their aims, methodology and criteria for making valid claims, approaches towards nature, and terminology. Ecology aims at understanding patterns and processes in the biosphere, relying on empirical research and statistics (quantitative methods). Legislation aims at regulating human behaviour in order to conserve nature. Criteria for making valid claims are legitimacy, logical consistency and feasibility in terms of providing clear procedures (qualitative methods).

Protected species are selected based on ecological, societal, ethical and aesthetic criteria. Prohibition provisions of species protection (FFA) forbid removal of protected plants, and disturbance of individual animals and parts of their habitat. Dispensation can be obtained when offence of the provisions is not expected to lead to negative effects on species populations. Regulations for area protection (BHD) intend to insure maintenance of quantity and quality of protected areas in terms of their ecological function for (populations of) qualifying species and natural habitats (i.e., phyto-sociological units), and their contribution to the coherence of the European ecological network Natura 2000.

It is concluded that the ecological reality is much more complex than the legislator has implemented in the legal framework for nature conservation. Several recent ecological insights, such as the importance of ecosystem dynamics, heterogeneity, non-linear behaviour, and uncertainty, have not yet or only partially been implemented. This causes both too wide information needs and many too restricted criteria for determining negative effects.

The selection of protected species and legal procedures for their protection frequently are too limited from an ecological point of view. Conservancy legislation seems to aim mainly at large and appealing species. There are, for example, no representatives of taxonomic groups like lower plants and only a few macroinvertebrates are protected by national and international legislation. Some of these taxonomic groups may be highly relevant ecological indicators (***e.g.***, Van den Brink ***et al.***, 1996; Lenders ***et al.***, 2001). Nevertheless, birds and mammals often have, because of their broad ecological requirements, an umbrella function (Simberloff, 1998). Furthermore, in the case of river management, the results of chapter V make clear that measures aimed at protected and endangered river characteristic species will also be beneficial for the riverine species assemblage as a whole.

Definitions of many ecological terms in legislation are sometimes vague and they often deviate from generally accepted ones in ecological sciences. However, there is also still a lot of discussion among ecologists about the meaning of ecological concepts such as disturbance, stress, and key-stone species. Therefore, strict definitions of ecological terms in the legal framework are not possible.

Consequences for river management

Traditional river management aimed at maintenance of structural and functional status quo in river systems. The negative consequences of this approach for biodiversity were described in chapter I. New river management strategies (combining nature development and flood risk reduction) aim at increasing river dynamics in floodplains. This new approach collides with the philosophy of nature legislation, which is oriented towards conservation.

Ecosystem and landscape dynamics, heterogeneity and non-linear behaviour are crucial aspects for the management of river floodplain ecosystems, as shown in chapter I and VI. In addition, uncertainty plays an important role in effect prediction. Therefore, the minor attention for these aspects in legislation can be problematic. However, within the legal framework there are various opportunities for including these aspects in the management of river floodplain ecosystems.

Possibilities for integration of ecological knowledge with legal instruments for rivers

Due to the differences in aims and methodology, the ecological and legal framework cannot completely be integrated. However, ecological knowledge and legal instruments can be attuned and combined in a pragmatic manner. This kind of integration has a partial character. In order to make legal instruments more suitable within the context of river management, key processes such as hydrodynamics, morphodynamics, grazing, and succession must be given more attention in the interpretation of the regulations in legislation.

The legal framework provides possibilities for dealing with dynamics, heterogeneity and uncertainty in legal management plans, codes of conduct, jurisprudence, and quality standards for ecological effect assessment. Opportunities for bridging the gap are:

1. Drawing up management plans for protected areas (SACs) in which dynamics such as inundation and succession, and landscape heterogeneity, are part of the conservation and management objectives.

2. Ensuring that the codes of conduct within the framework of the FFA are well underpinned by ecological knowledge and include crucial (river-specific) life-history traits of species: habitat aspects, tolerance levels and (meta)population structure and functioning.
3. Paying more attention within legal procedures to controlling (key) processes (environmental dynamics, succession and exchange of genes), functioning of ecological networks, and life-history traits. Dutch jurisprudence already mentioned acreage and connectivity of species' habitats, corridor function and migration of species, tolerance levels, metapopulation aspects, habitat heterogeneity and uncertainty. Jurisprudence regarding river floodplains in the Netherlands has included dynamics of sediments and habitat types.
4. Providing quality standards for ecological effect assessment. These quality standards could include crucial species-specific environmental factors and response variables (e.g. what does the species depend on, what are its tolerance limits and what is it capable of?). Effect assessment must take place at the most relevant level, which is usually the regional metapopulation network. Assessment of effects on habitats of species must include foraging and migration habitat, and spatio-temporal aspects. Population studies should pay attention to population structure and dynamics. Clear procedures for dealing with uncertainty in effect assessments must be designed, in which a certain degree of stochasticity is recognised as real and quantifiable phenomenon.

Opportunities for improving the ecological framework

Many problems in the practice of legal procedures and nature conservation relate to limitations of the state of the art of ecological science. More insight is required concerning the distribution of protected species and important species they depend on, their habitat (the relation between organism and environment), chaotic and non-linear phenomena, and the response of these species and ecosystems to human activities. Cumulative impacts should be studied more thoroughly. There are large uncertainties concerning response of individuals, populations and ecosystems to human activities, which must be quantified. Species specific standards for the conservation of metapopulations must be determined (e.g. required probability of metapopulation viability over a period of 100 years; cf. Verboom ***et al.***, 2001). Consensus is required about the ecological relevance of species (e.g. indicator, umbrella and keystone species) and the meaning of many ecological terms. Ecological knowledge should be better disseminated by means of easy accessible (web-based) databases. In order to build valid models, an experimental approach to impact studies could be adopted in which for several activities (1) the initial situation is determined, (2) the developments after human interventions are monitored and compared to developments in reference areas so that causality and dose-response relations can be analysed, and 3) insight is gained in possibilities for extrapolation of known dose-response relations to other species, areas and types of measures.

BIO-SAFE is a way to implement the species approach in nature conservation; however, also the protection of areas (area approach) must be accounted for. Models must be developed that can describe the natural characteristics of protected areas in physical, chemical and biological terms. Assessment models should also link species directly to the main abiotic factors (Pelsma, 2002; Aggenbach & Pelsma, 2005). Of course, one single model can never account for all legally relevant information and all aspects of species and ecosystems. Therefore, models must be able to establish input-output relations.

4.2 Integral river management

In the last paragraph, it was shown that combining and mutually attuning two entirely different frameworks (i.e. ecology and legislation) is possible. Throughout this thesis it was also shown that this kind of integration is a very difficult process and can take place to a limited extent only. It is important to note that the integration presented in this thesis concerns only two of the many relevant aspects in river management. In this paragraph, the results of this thesis are placed in the broader context of integral river management. This analysis answers research question 7 of chapter I: What is the contribution of the results of this thesis to integration in river management? In order to do this, we must first look at what integral river management means in current practice and what it theoretically could mean.

Integral water management has in the Netherlands been used as the term for managing water systems in a way that ensures taking all relevant aspects concerning water management into account (Dutch Ministry of Transport, Public Works and Water Management, 1998). This integrated water management began in the Netherlands around the mid eighties with the policy document ‘Dealing with water’ (Dutch Ministry of Transport, Public Works and Water Management, 1985). Integration in water management is the result of the water system approach in which the total system of biotic and abiotic elements of a certain water environment is taken into account (Van Ast, 1999). In this definition however, the human dimensions of water systems are neglected.

Environmental governance can be defined as a body of values and norms that guide or regulate state-civil society relationships in the use, control and management of the natural environment. These norms and values are expressed in a complex chain of rules, policies and institutions that constitue an organisational mechanism through which both broad objectives and the specific planning targets of environmental management must be articulated (Mugabe & Tumushabe, 1999).

Integral river management concerns physical, chemical, biological, landscape-scenic (aesthetical), cultural-historical, economical, social, legal, ethical, and psychological aspects of river systems. These aspects relate (amongst others) to the values safety, chemical quality, nature conservation, landscape quality, cultural-historical integrity, prosperity, equity, justice, legitimacy and well-being, respectively (Appleton, 1975; De Groot, 1992; Schama, 1995; Costanza ***et al.***, 1998; De Groot, 1998; Van Geest & Hödl, 1998; Hendrikx, 1999; Swart ***et al.***, 2001; Wilber, 2001a,b; Lenders, 2003; Walter & Shrubsole, 2003; Brouwer & Van Ek, 2004; Esbjörn-Hargens, 2005).

According to integral theory (Wilber, 2001a,b), all individual, social and environmental phenomena can be classified into four quadrants. These quadrants are four distinct dimensions of reality, or four ways of looking at the same occurrence. They are given by the interior (values, perceptions, etc.) and exterior (behaviour, empirical traits, etc.) of individuals and collectives (Figure 3). The quantitative, empirical aspects of objects are found on the right hand side, with individual aspects in the upper right and systems in the lower right quadrant. Qualitative, interpretative aspects are found on the left hand side, with personal values (psychological) in the upper left and collective values (cultural) in the lower left quadrant.

Apart from quadrants, this way of representing reality also uses levels, which represent levels of complexity and integration. These levels range from more fundamental (e.g. physical) to more significant (e.g. mental), as Wilber (2001b) puts it. The four quadrants are a simple way to organize the countless subjective and objective dimensions of individuals, societies and the environment. These dimensions have been intensely investigated by literally hundreds of major paradigms, practices, methodologies, and modes of inquiries (Wilber, 2001b).

When this classification is applied to river systems, behaviour and traits of individual water and sediment particles, chemicals, species, and people, are placed in the upper right quadrant. Patterns and processes of geohydrological systems, chemical systems, ecosystems, and the quantitative aspects of socio-economical systems and legal systems are placed in the lower right quadrant. Collective values concerning river systems, like safety against flooding, chemical quality of water and soil, ecological quality, scenic quality and cultural-historical integrity of the riverine landscape, prosperity and equity within the socio-economic situation of river-related regions, and justice and legitimacy of for example river management, are found in the lower left quadrant.

Each aspect on the left hand side of the figure has its main correlate on the right hand side. For example, the structure and behaviour of geohydrological systems (e.g. amounts of water and sediment moving through a winterbed) is most directly related to the safety aspect of river management. Prosperity and equity are most directly related to patterns and processes in socio-economical systems. Furthermore, aspects in the upper quadrants are always part of the collectives of the lower quandrants. Species are part of ecosystems, personal values and perceptions (psychological aspects) related to safety, health, identity and binding with the landscape, and well-being are always embedded in a cultural context. Figure 3 schematically represents only a few aspects and relations between these aspects within the highly complex reality of river systems and their inhabitants. A more detailed discussion is outside the scope of this thesis.

Integral theory is a means to take into account all aspects of a given part of reality and to organise these aspects in a way that provides insight into their relations and possibilities for integration in a larger conceptual and methodological framework. It also helps to appreciate the myriad of aspects and the methodologies used to study and govern them.

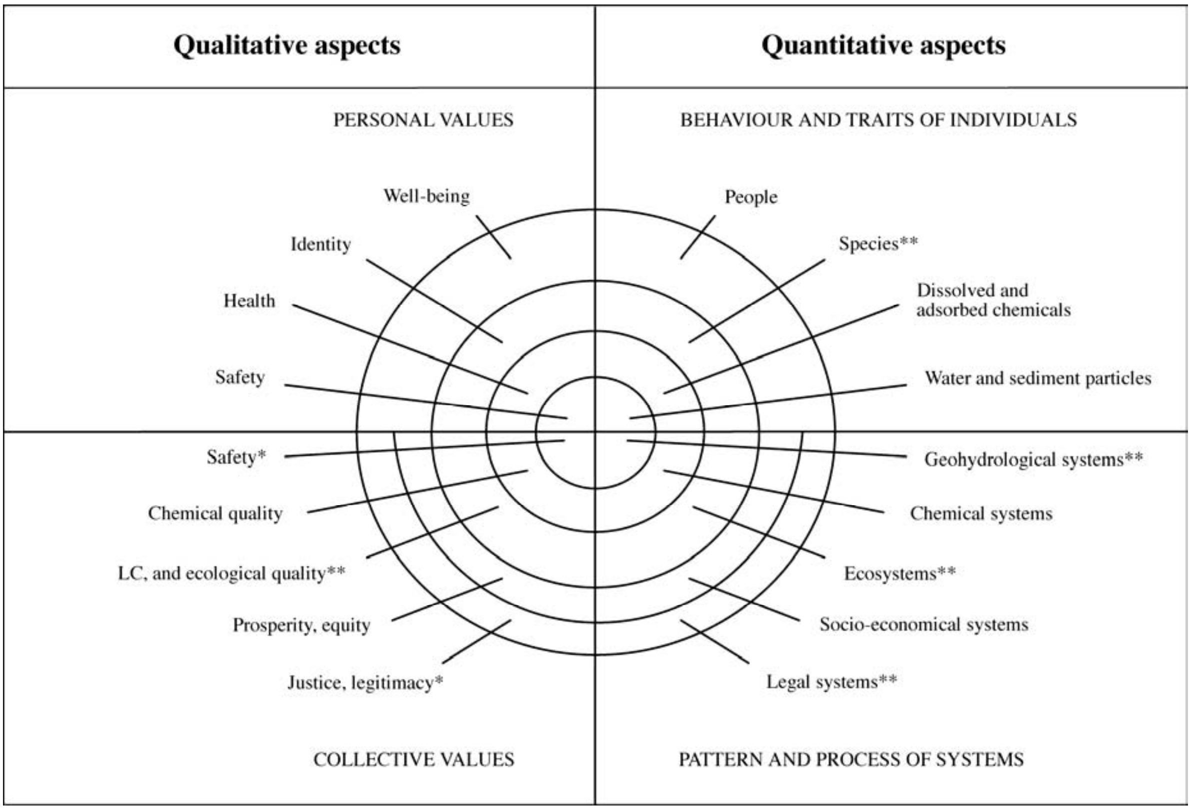

Figure 3. Qualitative and quantitative aspects of river management on various levels of complexity and integration. Adapted from Wilber (2001b). LC: Landscape quality and Cultural-historical integrity. * = referred to in this thesis; ** (partly) included in BIO-SAFE

The contribution of this thesis to integral river management

As mentioned in the introduction of this synthesis, Rotmans ***et al.*** (1996) define integrated assessment as: an interdisciplinary process of combining, interpreting and communicating knowledge from different scientific disciplines, in a way that provides useful information to decision makers. According to Parson (1995), assessment is defined by its purpose of assembling, summarizing, organising, interpreting, and possibly reconciling pieces of existing knowledge, and communicating them so that they are relevant and helpful for the deliberations of an intelligent but inexpert policy maker.

The research presented in this thesis was aimed at exactly this kind of integrated assessment. Two entirely different frameworks for nature conservation were combined (albeit partially) and mutually adapted. This approach has been the basis for communication between ecology and legislation. Communication can show negative consequences of limitations of both frameworks, and the opportunities for improving them. Here lies one part of the contribution made by this thesis. The other contribution is the possibility offered by BIO-SAFE to integrate the two frameworks coherently in river management.

This thesis is concerned only with the relation between nature conservation legislation (in terms of protected and endangered river characteristic species) and river-floodplain reconstruction and management aimed primarily at safety and ecological rehabilitation. Information in the model concerns only the behaviour in the species as concerns their selection of river ecotopes as habitat. Patterns and processes of riverine ecosystems included the model BIO-SAFE are hydrodynamics, morphodynamics and management dynamics. The use of ecotopes enables the model to integrate geohydrological aspects with species and ecosystems. Assignment of values to species and ecotopes based on their legal status integrates legal aspects with ecological aspects. The limitations of this approach were described in paragraph 3.1 and 3.2.

Quantitative aspects of legal systems included in this thesis are the numbers of protected and endangered species, and the frequency of occurrence of ecological items in jurisprudence. Qualitative aspects dealt with relate to approaches towards nature and the definition of ecological terms within the legal framework as well as the ecological framework for nature conservation. Nature conservation itself is done for ecological, economical, societal, aesthetical and ethical reasons (Backes, 2004; Swart, 2001; see also chapter VI). In chapter I, ecological and political-legal approaches to valuation of species and ecosystems were discussed. An integral approach to valuation of nature could also include ecosystem services (Costanza ***et al.***, 1998), economical value (Nunes & Van den Bergh, 2001), societal value (Brouwer & Van Ek, 2004), psychological and aesthetic value (Appleton, 1975; Misgav, 2000).

The approach chosen in this thesis is explicitly aimed at applicability within a societal context. Models must be useful, but must also be able to criticise the status quo in policy and management, otherwise there is the risk that to much emphasis may be placed on making life easy for policy makers rather than on challenging their ways of working where these lead to decisions that threaten environmental sustainability (Scrase & Sheate, 2002).

What is still needed for integral river management?

In river management, many models are used, e.g. hydraulic models, ecotoxicological models and metapopulation models (Leuven & Poudevigne, 2002). These models must determine whether present situations or reconstruction designs meet political, legal, and socio-cultural goals concerning safety, ecotoxicological risks and nature conservation. In order to do so, models must produce quality standards concerning all these aspects. Quality standards and assessment models are also required for landscape quality, cultural historical identity, socio-economical aspects and legal aspects.

The knowledge that is available from the great variety of sciences, hydraulics, geophysics, water and soil chemistry, human risk assessment, ecotoxicology, river ecology, population dynamics, landscape ecology, economics, sociology and psychology, is integrated in river management to a limited extent. Historically, the disciplinary training of most professionals working in water management has been in the natural sciences or engineering. The dominant mindset has been one of positivism and determinism, which entails a disinterest in historical and socio-cultural context, and an emphasis on quantification and technical rationality (Scrase & Sheate, 2002). There is little attention for socio-cultural and psychological aspects of river floodplain ecosystems such as identity of the river landscape and well-being of humans who live there. Furthermore, integrative river management as currently conceived focuses almost exclusively on the quantitative system aspects of rivers (lower right of Figure 3).

For integral river management, more attention is required for aspects that are placed in the other three quadrants. However, integration among assessment models in order to take all aspects into account might be problematic. Differences between approaches often result in incompatibility. Nevertheless, in other respects models are often complementary and could be integrated in a consecutive, encompassing, or overlapping manner. In many cases, sequential application of the tools can be more productive than the tedious construction and application of the encompassing model that integrates many other models (Bouman ***et al.***, 2000).

Apart from giving attention to qualitative and individual aspects of river systems, river management should also give more attention to interaction and the supra-national dimension of river management. For the coming years, interaction can be considered as a new keyword. Interaction can be found in the mutual relationship between the water managers and the water systems they want to influence, and in the horizontal way agencies try to influence society. In other words: in integrated water management, citizens are given the possibility to express their opinion, but in interactive water management, people think together with the water managers about the most desired developments. Another remaining issue is whether, and when, the actual developments will lead to a consequent translation of water system management on a supra-national level (Van Ast, 1999).

In the end, the possibilities of policy makers and managers to govern river-floodplain systems determine the quality of these systems. Models can only give a modest contribution. Integrated approaches and integration are not in itself a solution to all problems in water management. The concept of integrated water management creates as many challenges as it might resolve (Scrase & Sheate, 2002). There obviously still is a lot of work to do.

5 Conclusions and recommendations

Based on the results presented in this synthesis, is concluded that:

1. BIO-SAFE is an operational and scientifically underpinned model for integration of ecological knowledge and information with legal instruments for nature conservation in river management that provides useful information in evaluation studies, is valid for impact assessments and insensitive to value judgements.
2. Measures taken in floodplains can have positive as well as negative effects on protected and endangered species, but when landscape heterogeneity is maximized and low-intermediate dynamic circumstances are conserved and developed by literally giving more room to rivers and/or creatively using available space, nature conservation can be combined with flood risk reduction as well as ecological rehabilitation.
3. The ecological and the legal frameworks for nature conservation have different aims, criteria for making valid claims and approaches towards nature and species selection,

but there are various opportunities for integration of ecological knowledge with legal instruments in for example effect studies, management plans and codes of conduct.
4. BIO-SAFE contributes to integration in river management by combining and mutually adapting ecological knowledge and legal instruments in an assessment model which provides useful information to decision makers.

The following recommendations are made:
1. BIO-SAFE can be optimised ecologically by using more detailed ecotope classifications, setting minimum required surface areas for species and populations, and algorithms that describe non-linear relationships between ecotope surface areas and potentials for species and taxonomic groups.
2. Optimisation concerning the legal aspects of BIO-SAFE should provide the possibility to assess impacts on the level of species, assess impacts on protected areas, and account for the various delicacies of the legal status of species.
3. BIO-SAFE should be used early in planning processes for designing reconstruction scenarios that optimise opportunities for protected and endangered biodiversity, during the fine-tuning of the design (just before execution of projects) in order to determine which ecotopes are in the actual situation the most valuable and should be conserved, and as a tool for evaluation of measures taken in river floodplain systems.
4. Integration in river management requires more attention for socio-cultural and psychological aspects of river systems, in particular aesthetics, identity and cultural-historical integrity of the riverine landscape, and perceptions and ideas among citizens and local governments. Furthermore, addressing the supra-national dimension of river management and creating a sound interaction between river managers and the river systems they want to influence, as well as the citizens that live there, are great challenges for the future.

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Summary

1 Background and goals

Legal instruments for nature conservation make knowledge on actual and potential presence of protected and endangered species in river floodplain ecosystems an important issue in river management. The goal of these legal instruments is the conservation and, where appropriate, the restoration of biodiversity. Ecological scientists often define biodiversity as the variability of living organisms and ecological complexes, and having three levels (genetic, species and ecosystem) and three components (compositional, structural and functional diversity). River systems have high biodiversity potential and are therefore important for the conservation and restoration of biodiversity.

Within river management, many physical reconstruction and management plans are currently being carried out and planned. Major goals are flood risk reduction, ecological rehabilitation and economic development. The expected effects on biodiversity are massive and can be positive as well as negative. Nature conservation legislation demands that these effects are thoroughly assessed, weighted against other interests, mitigated and compensated for if necessary. Huge amounts of ecological knowledge will be required to do the obligatory assessments, as well as the studies for underpinning river rehabilitation. Therefore, ecology and legislation are two crucial frameworks for nature conservation in river management. It is a great challenge to combine flood risk reduction, ecological rehabilitation and nature conservation in optimised packages of river reconstruction and management measures. This challenge calls for science-based models that can integrate ecological knowledge with legal instruments for nature conservation in river management.

In this thesis, such a model is developed, evaluated and applied. Application of the model shows the effects of river management measures on actual and potential presence of protected and endangered river species. It also shows the implications of choices made in the legal framework for the possibilities of river managers to design, create and manage our present and future riverine landscape. This thesis also studies the relation between ecological knowledge and legal instruments from a more theoretical point of view. Analysis of goals, approaches towards nature and species selection, and of terminology, provides useful insights into the possibilities and limitations for integration of the two crucial frameworks for nature conservation in river management.

Chapter I describes the relation between rivers and biodiversity, the influence of human activities on biodiversity, nature conservation policy and legislation, and different approaches to biodiversity valuation. This chapter also describes the delineation, goals and research questions of the study presented in this thesis.

2 Development and application of BIO-SAFE

Chapter II describes the development of an operational, transnational model for valuation of species, ecotopes, and areas, as well as for assessment of impacts of changes in river-floodplain ecosystems of the Rhine and Meuse, based on protected and endangered species. This model is called BIO-SAFE (Spreadsheet Application For Evaluation of BIOdiversity). BIO-SAFE was developed into a tool for biodiversity assessment with regard to design and evaluation of physical planning projects, Environmental Impact Assessments and comparative landscape-ecological studies. It was conceived to be applicable in Germany, France, Belgium and the Netherlands.

BIO-SAFE incorporates a selection of river characteristic species, mentioned in Red Lists, the European Birds and Habitats Directive, and the conventions of Bern and Bonn. Taxonomic groups involved are higher plants, birds, herpetofauna, mammals, fish, butterflies and odonates (dragon- and damselflies). Every species has been linked to one or more ecotopes (e.g. natural grasslands, softwood forest, side channel) based on its habitat preferences. Because every species is also assigned a value based on its status according to the Red Lists, European Directives and international conventions, areas, ecotopes and reconstruction and management plans can be assessed in terms of protected and endangered species. The model requires input data on presence of species and/or surface area of ecotopes.

The model has been applied to flood risk reduction projects along the rivers Rhine and Meuse. Results show that BIO-SAFE yields quantitative information regarding the degree to which actual situations, reconstruction designs and developments of species and ecotope compositions meet national and international agreements on nature conservation.

3 Complementarity and indicator function of BIO-SAFE

Chapter III focuses on the possibilities for evaluation of developments of species compositions after execution of rehabilitation measures in floodplains. Furthermore, the degree to which BIO-SAFE yields complementary and/or indicative information compared to a conventional valuation method is studied. This chapter evaluates the effects of ecological rehabilitation on biodiversity in floodplains along lowland rivers in the Netherlands, using two different approaches. Approach 1 uses species richness (a conventional method), approach 2 uses BIO-SAFE.

In most cases, biodiversity in the floodplains significantly increased after rehabilitation. However, the outcome of the evaluation differs per taxonomic group and approach. The results show positive temporal trends with respect to species richness of higher plants, herpetofauna and odonates ($p<0.0083$), and to a lesser degree mammals and butterflies ($p<0.05$). The BIO-SAFE index increased to some extent, but not significantly, for higher plants, herpetofauna and butterflies ($p<0.05$). Correlation between species richness and model output was found in all sites for birds, herpetofauna, odonates ($p<0.0083$) and, to a lesser degree, higher plants ($p<0.05$). For higher plants, positive as well as negative correlations were found, depending on the site.

Based on these results, we argue that (1) the rehabilitation measures studied are only partly successful and need optimisation, particularly for birds and protected and endangered mammals and odonates; (2) BIO-SAFE gives useful complementary information for higher plants, mammals and butterflies; (3) the model appears to be indicative for developments of species richness of birds, herpetofauna and odonates.

4 Validity and sensitivity of BIO-SAFE

The application of the model in scenario analysis and impact assessment is studied in chapter IV. This chapter investigates the validity of prediction of effects planned measures using BIO-SAFE, and the sensitivity of the model to assignment of values to species and ecotopes. The validity of BIO-SAFE has been tested by comparing effects of landscape changes predicted by the model on the diversity of protected and endangered species with observed changes in species diversity in five reconstructed floodplains. The sensitivity of BIO-SAFE to the assignment of values to species and ecotopes has been analysed as follows: the weights given to different valuation criteria for protected and endangered species, i.e. EU Birds and Habitats Directive, the conventions of Bern and Bonn and Red Lists, were varied and the effects on ranking of alternatives were quantified. The alternatives for reconstruction were taken from a

Strategic Environmental Assessment concerning the Spatial Planning Key Decision 'Room for the River' for reconstruction of the Dutch floodplains of the river Rhine, aimed at flood risk reduction, improvement of spatial quality and ecological rehabilitation. Examples of measures are lowering floodplains, shifting dikes away from the river or creation of retention basins for high water peaks.

A statistically significant correlation ($p < 0.01$) was found between effects predicted by the model based on ecotope data and observed effects calculated based on changes in species composition. The sensitivity of the model to value assignment proved to be very limited. Comparison of five realistic valuation options (e.g. 'only Red Lists are taken into account', 'all valuation criteria are weighed equally') showed that different rankings of scenarios predominantly occur when valuation criteria are left out of the assessment. Based on these results we conclude that BIO-SAFE is a valid model for impact assessments.

Quantification of sensitivity of impact assessment to value assignment shows that a model like BIO-SAFE is not very sensitive to assignment of values to different policy and legislation based criteria. Arbitrariness of the value assignment therefore has a very limited effect on assessment outcomes. However, the decision to include valuation criteria or not is very important.

5 Protected and endangered river species and hydodynamics

In chapter V, BIO-SAFE is applied for valuation of ecotopes on the basis of their importance to protected and endangered river characteristic species. This is used to give recommendations concerning the realisation of nature conservation goals. Because ecotopes are also defined by the hydrodynamic conditions under which they occur (i.e. number of days flooded per year), BIO-SAFE can be used to quantify the relation between protected and endangered river species and different hydrodynamic conditions in floodplains. This combines ecological and policy-legal aspects of biodiversity in river management. The importance of different hydrodynamic conditions along a lateral gradient (varying from always flooded to almost never flooded) was quantified for various taxonomic groups.

The results show that (1) protected and endangered river species require ecotopes along the entire hydrodynamic gradient; (2) different parts of the hydrodynamic gradient are important to different species, belonging to different taxonomic groups; (3) intermediate and low-dynamic parts are particularly important for protected and endangered species and (4) these species differ in their specificity for hydrodynamic conditions. Many species of higher plants, fish and butterflies have a narrow range for hydrodynamics and many species of birds and mammals use ecotopes along the entire gradient.

Patterns calculated with the model BIO-SAFE match very well with patterns observed by other researchers concerning species spectra that are much broader than protected and endangered species only. Even when focussing only on protected and endangered river species, the entire natural hydrodynamic gradient is important. This means that the riverine species assemblage as a whole can benefit from measures focussing on protected and endangered river species only. River reconstruction and management should aim at re-establishing the entire hydrodynamic gradient, increasing the spatial heterogeneity of hydrodynamic conditions, and conserving low and intermediate dynamic conditions. This result criticises the current way of thinking in river management, which emphasises increasing river dynamics in floodplains and removing low dynamic ecotopes. Optimisation of reconstruction measures in terms of protected and endangered species calls for creative spatial solutions which will frequently require space in the lateral dimension.

6 Relating ecology and legislation in river management

In chapter VI the relation between ecological theory and practice and legal instruments for nature conservation is analysed and linked to river management practice. Ecological knowledge has been implemented in legislation to protect species and their habitats, and the two frameworks meet in the practice of nature conservation in legal procedures that require ecological information. However, differences in approaches towards nature, and terminology, cause several problems. There is a lack of clarity about legal obligations concerning required ecological information as well as the extent of ecological effect assessment. Moreover, from an ecological-scientific perspective nature conservation legislation and jurisprudence often seems illogic and incomplete. Conversely, ecological science cannot always provide all the answers to legal questions, nor the certainty that is required in legal procedures.

It is concluded that ecological reality is much more complex than the legislator has implemented in current legislation. Several recent ecological insights have not yet or only partially been implemented (such as the importance of ecosystem dynamics, heterogeneity, non-linear behaviour and uncertainty). This causes both enormous information needs and many too restricted criteria for determining effects on species and ecosystems. Especially the selection of protected species and legal procedures for their protection are too limited from an ecological point of view. This leaves important potential effects out of the scope of legal procedures. Definitions of many ecological terms in legislation are sometimes vague and they often deviate from generally accepted ones in ecological science. In many cases however, the differences are inevitable. Legislation needs to insure that a clear framework for maintaining law and order exists, which is internally consistent and provides legal security. Implementation of all ecological insights, that are developing continuously, will almost certainly come into conflict with the abovementioned criteria. Nevertheless, possibilities for bridging the gaps can be searched for.

In order to make legislation more appropriate for nature conservation and ecological research more relevant for legal procedures, more attention must be paid to the ecological relevance and extinction risk of species and ecosystems. More attention must also be paid to ecosystem dynamics (including key processes such as the influence of organisms on chemistry of soil and water, environmental dynamics, succession and exchange of genes), ecological scale, spatial heterogeneity, and structure and function of ecological networks. Furthermore, legislation should anticipate more appropriate on uncertainties in effect assessments. Opportunities for bridging the gaps between ecology and legislation are designing ecologically sound management plans that must be drawn up for protected areas, and codes of conduct for obtaining exemption from the Flora and Fauna Act. Also jurisprudence is an important way to fill gaps in legislation, and offers concrete possibilities for improving input of ecological knowledge and information in legal procedures. Last but not least, quality standards for ecological effect assessments can be provided. Effect assessments should pay more attention to cumulative effects and should consider the most relevant level, which is usually the regional metapopulation network. Within the ecological field, more insight is required into the distribution of protected species, their habitat, ecological networks, and the response of species and ecosystems to human activities. In addition, more consensus about ecological relevance of species and the definition of many ecological terms would be highly valuable.

The jurists and decision makers who apply the current nature conservation legislation should try and understand more of an ecological approach to nature conservation. Ecologists should realise that legislation represents a world of thought in its own right, with its own objectives (e.g. offering an ethical framework for dealing with nature, legal security and maintainability) and criteria for making valid claims (like internal consistency and logical fit).

7 Synthesis and conclusions

In the synthesis (chapter VII) the possibilities and limitations of the model are summarized and recommendations for improvement are given. Furthermore, the implications of nature conservation legislation for reconstruction and management, the effects of river management measures on protected and endangered species as well as the opportunities for integration of ecological knowledge with legal instruments for nature conservation in river management are discussed.

Based on the results of chapter II to VI it is concluded that BIO-SAFE is an operational and scientifically underpinned model for integration of ecological knowledge and information with legal instruments for nature conservation in river management, that provides useful information in evaluation studies, is valid for impact assessments and insensitive to value judgements. Important ecological limitations of the model concern genetic and population aspects (which are not included), habitat description (which is based only on ecotopes), and effect prediction (based only on species-ecotope links and relative surface area of ecotopes). There are also legal limitations of the model. BIO-SAFE is useful only in some parts of the legal procedures for nature conservation, includes only river characteristic species, and does not assess effects on the level of separate species. Other legal limitations concern value assignment, which does not always correspond with legislation.

The results of model application in chapter II to V concern the importance of ecotopes, the effects of measures taken in floodplains, and impact assessment. Ecotope valuation shows that every taxonomic group is related to different sets of ecotopes and that no ecotopes and parts of the hydrodynamic gradient can be neglected when protected and endangered species are concerned. Low and intermediate dynamic ecotopes are particularly valuable. Now that river management emphasises increasing dynamics, we should stress that intermediate and low dynamic ecotopes are important. Rehabilitation and reconstruction measures in floodplains show positive as well as negative effects. Overall, the positive effects are promising but limited. Impact assessment shows that the most natural situation as envisaged by planners, does not necessarily correspond with the highest potential for protected and endangered river species. Scenarios that increase river dynamics in floodplains are in some cases expected to lead to negative or suboptimal effects.

The measures taken can be optimised by increasing the spatial scale, and using detailed knowledge on local potentials. When landscape heterogeneity is maximized and low and intermediate dynamic circumstances are conserved and developed, nature conservation can be combined with flood risk reduction and ecological rehabilitation. This will often require space in the lateral dimension and/or creative use of available space.

The ecological and the legal framework for nature conservation have different aims, criteria for making valid claims and approaches towards nature and species selection. However, there are various opportunities for bridging the gap between ecological knowledge and legal instruments in practice, e.g. in jurisprudence, management plans, codes of conduct and effect studies. BIO-SAFE contributes to integration in river management by combining and mutually adapting ecological knowledge and legal instruments in an assessment model which provides useful information to decision makers.

It is recommended that BIO-SAFE be optimised ecologically by using more detailed ecotope classifications, setting minimum required surface areas for species and populations, and algorithms that can describe non-linear relationships between ecotope surface areas and potentials for species and taxonomic groups. Optimisation concerning the legal aspects of BIO-SAFE should provide the possibility to assess impacts on the level of separate species, and account for the various delicacies of the legal status of protected species. BIO-SAFE is best used early in planning processes for designing reconstruction scenarios that optimise

opportunities for protected and endangered biodiversity, during the fine-tuning of designs in order to determine which ecotopes are in the actual situation the most valuable and should be conserved, and as a tool for evaluation of measures taken in river floodplain systems.

Integration in river management requires more attention for socio-cultural and psychological aspects of river systems, in particular scenic quality, identity and cultural-historical integrity of the riverine landscape, and perceptions among citizens and local governments. Furthermore, addressing the supra-national dimension of river management and creating a sound interaction between river managers and the river systems they want to influence, as well as interaction between river managers and the citizens that live there. Integration and interaction in river management are great challenges for the future.

Samenvatting

1 Achtergronden en doelstelling

Wettelijke instrumenten voor natuurbescherming maken dat kennis over actuele en potentiële aanwezigheid van de beschermde en bedreigde soorten in rivierecosystemen een belangrijk onderwerp is in het rivierbeheer. Het doel van wettelijke instrumenten voor natuurbescherming is het behoud en, waar mogelijk, het herstel van biodiversiteit. Ecologische wetenschappers definiëren biodiversiteit wel als de variabiliteit van levende organismen en ecologische complexen, opgebouwd uit drie niveaus (genen, soorten en ecosystemen) en drie componenten (diversiteit in compositie, structuur en functioneren). Rivierecosystemen worden gekenmerkt door een hoog biodiversiteitpotentieel en zijn daarom van groot belang voor het behoud en herstel van biodiversiteit.

Binnen het rivierbeheer wordt momenteel een groot aantal plannen voor herinrichting en wijziging van beheer opgesteld en uitgevoerd. Belangrijke doelen zijn het verlagen van overstromingsrisico's, ecologisch herstel en economische ontwikkeling. De te verwachten effecten op biodiversiteit zijn ingrijpend en kunnen zowel positief als negatief uitpakken. Natuurbeschermingswetgeving vereist dat deze effecten zorgvuldig worden beoordeeld, afgewogen tegen andere belangen, verzacht en gecompenseerd als dat nodig is. Enorme hoeveelheden ecologische kennis en informatie zullen nodig zijn om deze verplichte beoordelingen, en het onderzoek dat nodig is voor wetenschappelijke onderbouwing van maatregelen voor ecologisch herstel, uit te voeren. Daarom zijn ecologie en wetgeving twee cruciale kaders voor natuurbescherming in het rivierbeheer. Het is een grote uitdaging om het verlagen van overstromingsrisico's, ecologisch herstel en natuurbescherming te combineren in geoptimaliseerde maatregelenpakketten. Deze uitdaging vraagt om wetenschappelijke modellen die ecologische kennis kunnen integreren met wettelijke instrumenten voor natuurbescherming in het rivierbeheer.

In dit proefschrift wordt een dergelijk model ontwikkeld, geëvalueerd en toegepast. Toepassing van het model laat zien wat de effecten zijn van de rivierbeheermaatregelen op actuele en potentiële aanwezigheid van beschermde en bedreigde, voor het rivierengebied karakteristieke flora en fauna. Het laat ook zien wat de implicaties zijn van in het juridische kader gemaakte keuzen voor de mogelijkheden van rivierbeheerders om ons huidige en toekomstige rivierenlandschap te ontwerpen, vorm te geven en te beheren. Dit proefschrift bestudeert ook de relatie tussen ecologische kennis en wettelijke instrumenten vanuit een meer theoretisch perspectief. Analyse van doelen, de wijze waarop soorten en ecosystemen worden benaderd en gehanteerde begrippenkaders in ecologie en wetgeving levert waardevolle inzichten in de mogelijkheden en beperkingen voor integratie van de twee cruciale kaders voor natuurbescherming in het rivierbeheer.

Hoofdstuk 1 beschrijft de relatie tussen rivieren en biodiversiteit en de invloed van mensen hierop, natuurbeleid en wetgeving ten aanzien van natuurbescherming, en verschillende benaderingen voor het waarderen van biodiversiteit. Hierna geeft het hoofdstuk de afbakening en doel- en vraagstelling van het onderzoek.

2 Ontwikkeling en toepassing van BIO-SAFE

Hoofdstuk 2 beschrijft de ontwikkeling en toepassing van een operationeel en transnationaal model voor waardering van soorten, ecotopen en gebieden en voor beoordeling van veranderingen in rivierecosystemen van de Rijn en de Maas, gebaseerd op beschermde en bedreigde soorten. Dit model heet BIO-SAFE: Spreadsheet Application For Evaluation of

BIOdiversity. BIO-SAFE is ontwikkeld tot een instrument voor biodiversiteitwaardering in het kader van het ontwerpen en evalueren van herinrichtingprojecten, milieu-effectrapportages en landschapsecologische studies. Het model is op zo'n manier ontwikkeld dat het toepasbaar is in Duitsland, Frankrijk, België en Nederland.

BIO-SAFE bevat een selectie van rivierkarakteristieke soorten die worden genoemd in de Rode Lijsten, de Europese Vogel- en Habitatrichtlijn en de conventies van Bern en Bonn. De selectie betreft de taxonomische groepen hogere planten, vogels, herpetofauna (reptielen en amfibieën), zoogdieren, vissen, vlinders en odonata (libellen en waterjuffers). Elke soort is op basis van zijn habitatvoorkeuren gekoppeld aan één of meerdere ecotopen (bijvoorbeeld natuurlijke graslanden, zachthout ooibos, nevengeulen). Omdat aan elke soort bovendien een waarde is toegekend op basis van zijn status volgens de Rode Lijsten, Europese richtlijnen, en internationale conventies, kunnen gebieden, ecotopen en plannen worden gewaardeerd in termen van beschermde en bedreigde riviersoorten. Het model gebruikt invoergegevens over aanwezigheid van soorten en/of oppervlakte van ecotopen.

Het is toegepast op beheer- en herinrichtingsprojecten in de uiterwaarden van de Rijn en de Maas. De resultaten laten zien dat BIO-SAFE bruikbare, kwantitatieve informatie oplevert betreffende de mate waarin actuele situaties, herinrichtingontwerpen en ontwikkelingen van soorten- en ecotopensamenstellingen beantwoorden aan nationale en internationale overeenkomsten op het gebied van natuurbehoud.

3 Complementariteit en indicatorfunctie van BIO-SAFE

Hoofdstuk 3 focust op de mogelijkheden die BIO-SAFE biedt voor evaluatie van ontwikkelingen van soortensamenstellingen nadat ecologische herstelmaatregelen in uiterwaarden zijn genomen. Bovendien wordt de mate waarin BIO-SAFE aanvullende en/of indicatieve informatie oplevert ten opzichte van een conventionele waarderingsmethode onder de loep genomen. De effecten van ecologisch herstel op biodiversiteit in uiterwaarden langs de laaglandrivieren in Nederland worden beoordeeld volgens twee verschillende benaderingen. Benadering 1 gebruikt soortenrijkdom (een conventionele methode), benadering 2 gebruikt BIO-SAFE.

In de meeste gevallen neemt de biodiversiteit in de uiterwaarden significant toe nadat is begonnen met uitvoering van herstelmaatregelen. Echter, de resultaten verschillen per taxonomische groep en benadering. Significante toename van soortenrijkdom werd waargenomen voor de groepen hogere planten, herpetofauna en odonata ($p<0{,}0083$), en in beperkte mate voor zoogdieren en vlinders ($p<0{,}05$). De BIO-SAFE index nam alleen toe voor hogere planten, herpetofauna en vlinders, en dat slechts in beperkte mate ($p<0{,}05$). Correlatie tussen soortenrijkdom en uitkomsten van BIO-SAFE werd gevonden in alle gebieden voor vogels, herpetofauna, odonata ($p<0{,}0083$) en in beprekte mate voor hogere planten ($p<0{,}05$). Voor hogere planten werden zowel positieve als negatieve correlaties gevonden, afhankelijk van het gebied.

Uit deze resultaten wordt geconcludeerd dat (1) de bestudeerde maatregelen voor ecologisch herstel maar gedeeltelijk succesvol zijn, en vooral voor vogels en beschermde en bedreigde zoogdieren en odonata nog optimalisatie nodig hebben; (2) BIO-SAFE voor de groepen hogere planten, zoogdieren en vlinders nuttige aanvullende informatie oplevert; en (3) het model voor de groepen vogels, herpetofauna en odonata indicatief lijkt te zijn voor ontwikkelingen in soortenrijkdom.

4 Validiteit en gevoeligheid

Hoofdstuk 4 beschrijft de toepassing van het model in scenariostudies en effecten-beoordelingen die voorafgaan aan uitvoering van maatregelen. Tevens wordt dieper ingegaan op de validiteit van de effectvoorspelling met BIO-SAFE en de gevoeligheid van het model voor de toekenning van waarden aan soorten en ecotopen. De validiteit van het model is getest door middel van het vergelijken van met het model voorspelde effecten van herinrichtingmaatregelen in vijf rivieruiterwaardgebieden met daadwerkelijk waargenomen veranderingen in diversiteit van beschermde en bedreigde soorten. De gevoeligheid van BIO-SAFE voor toekenning van waarden aan soorten en ecotopen is als volgt geanalyseerd. De waarden die toegekend zijn aan de verschillende waarderingscriteria voor beschermde en bedreigde soorten (Vogel- en Habitatrichtlijn, de conventies van Bern en Bonn en de Rode Lijsten) werden gevarieerd en de gevolgen hiervan voor de beoordeling van verschillende alternatieven voor herinrichting van uiterwaarden werden gekwantificeerd. Deze alternatieven voor herinrichting waren afkomstig uit een strategische milieubeoordeling betreffende de Planologische Kernbeslissing ‘Ruimte voor de Rivier’.‘Ruimte voor de Rivier’ is een project herinrichting van de Nederlandse uiterwaarden van de rivier de Rijn, en benedenstroomse delen van de Maas dat is gericht op verlaging van overstromingsrisico’s, verbetering van ruimtelijke kwaliteit en ecologisch herstel. Voorbeelden van maatregelen zijn het verlagen van uiterwaarden, het landinwaarts verleggen van dijken of door het bestemmen van gebieden die bij hoogwater kunnen dienen om water tijdelijk op te vangen.

Er bleek een statistisch significante correlatie ($p<0,01$) te bestaan tussen door het model op basis van ecotoopgegevens voorspelde en op basis van waargenomen soortensamenstelling berekende waarden voor beschermde en bedreigde soorten. De gevoeligheid van het model voor waardetoekenning bleek zeer beperkt. Vergelijking van vijf realistische opties voor waardetoekenning (bijvoorbeeld ‘alleen Rode Lijsten tellen mee’, ‘alle waarderingscriteria wegen even zwaar’) liet zien dat afwijkende rangordes van scenario’s vooral optreden als waarderingscriteria worden weggelaten in de beoordeling. Op basis van deze resultaten is geconcludeerd dat BIO-SAFE valide is voor effectenbeoordeling.

Het kwantificeren van de gevoeligheid van effectenbeoordeling voor waardetoekenning toont aan dat een model zoals BIO-SAFE in zeer beperkte mate gevoelig is voor het toekennen van waarden aan de verschillende, uit beleid en wetgeving afgeleide, criteria. Eventuele subjectiviteit van de experts die de waarden hebben toegekend heeft daarom een zeer beperkt effect op de uitkomsten van de effectenbeoordeling. De beslissing criteria mee te nemen of niet, heeft echter grote gevolgen.

5 Beschermde en bedreigde riviersoorten en hydrodynamiek

In hoofdstuk 5 wordt BIO-SAFE toegepast voor het waarderen van ecotopen op basis van hun belang voor beschermde en bedreigde riviersoorten. Hiermee worden aanbevelingen gedaan aangaande het realiseren van doelen van natuurbescherming. Omdat ecotopen onder andere worden gedefinieerd door de hydrodynamische omstandigheden (aantal dagen overstroomd per jaar) waaronder ze voorkomen, kan met BIO-SAFE de relatie tussen beschermde en bedreigde riviersoorten en verschillende hydrodynamische omstandigheden in rivieruiterwaarden worden onderzocht. Op deze wijze worden ecologische en bestuurlijk-juridische aspecten van biodiversiteit in het rivierbeheer met elkaar gecombineerd. Het belang van verschillende hydrodynamische omstandigheden langs een gradiënt (variërend van altijd overstroomd tot vrijwel altijd droog) werd voor de verschillende taxonomische groepen in het model gekwantificeerd.

De resultaten laten zien dat (1) beschermde en bedreigde riviersoorten ecotopen langs de hele hydrodynamische gradiënt nodig hebben; (2) verschillende onderdelen van de hydrodynamische gradiënt van belang zijn voor steeds weer andere soorten uit verschillende taxonomische groepen; (3) middelmatig en laagdynamische onderdelen in het bijzonder belangrijk zijn voor beschermde en bedreigde soorten en (4) de soortgroepen sterk verschillen wat betreft hun amplitudo voor hydrodynamische omstandigheden. Veel soorten uit de groepen hogere planten, vissen en vlinders hebben een smalle amplitudo (gebruiken één of enkele ecotopen op een klein stukje van de gradiënt), veel vogel- en zoogdiersoorten gebruiken meerdere ecotopen langs de hele gradiënt.

De met het model BIO-SAFE berekende patronen komen zeer goed overeen met door andere onderzoekers waargenomen patronen voor soortenspectra die veel breder zijn dan alleen beschermde en bedreigde rivierkarakteristieke soorten. Zelfs wanneer alleen op beschermde en bedreigde rivierkarakteristieke soorten wordt gelet, is de gehele natuurlijke hydrodynamische gradiënt van belang. Dit betekent dat de biodiversiteit van rivieren in het algemeen gebaat is bij maatregelen die alleen gericht zijn op beschermde en bedreigde riviersoorten. Voor behoud en herstel van biodiversiteit dient herinrichting en beheer van rivieren zich te richten op het herstellen van de gehele van nature optredende hydrodynamische gradiënt, waarbij de ruimtelijke heterogeniteit van hydrodynamische omstandigheden wordt verhoogd, en laag en middelmatig dynamische omstandigheden worden behouden. Dit resultaat staat haaks op de huidige manier van denken in het rivierbeheer, waarbij sterk de nadruk ligt op een toename van rivierdynamiek in uiterwaarden en het verwijderen van laagdynamische ecotopen. Optimalisatie van maatregelen voor beschermde en bedreigde soorten vraagt om creatieve oplossingen. Hierbij zal vaak moeten worden gezocht naar ruimte in de breedte, en niet alleen in de diepte.

6 De relatie tussen ecologie en wetgeving in het rivierbeheer

Hoofdstuk 6 analyseert de relatie tussen ecologische theorievorming en onderzoek enerzijds en juridische instrumenten voor natuurbescherming anderzijds, en koppelt dit aan de praktijk van het rivierbeheer. Ecologische kennis is geïmplementeerd in wetgeving voor het beschermen van flora en fauna en hun habitat. Bovendien ontmoeten ecologie en wetgeving elkaar in de praktijk van natuurbescherming in juridische procedures waarin ecologische informatie nodig is. Verschillen in de wijze waarop natuur wordt benaderd, en tussen de gehanteerde begrippen en begripsdefinities veroorzaken verschillende problemen. Er is onduidelijkheid over de wettelijke verplichtingen betreffende de benodigde ecologische informatie en over vereiste reikwijdte en diepgang van ecologische effectenbeoordeling. Bovendien lijken wetgeving en de jurisprudentie vanuit een ecologisch perspectief vaak onlogisch en incompleet. Aan de andere kant kan de ecologische wetenschap vaak geen antwoord geven op alle juridische vragen, en ook niet de mate van zekerheid bieden die benodigd is in de juridische procedures.

Uit de analyse wordt geconcludeerd dat de huidige implementatie van ecologische kennis in de wetgeving geen recht doet aan de complexiteit van de ecologische werkelijkheid. Verschillende recente ecologische inzichten zijn nog niet of slechts gedeeltelijk geïmplementeerd. Voorbeelden zijn het belang van de dynamiek van ecosystemen, heterogeniteit, non-lineair gedrag en stochasticiteit (onzekerheid). Dit leidt er enerzijds toe dat de wetgeving de ecologie overvraagt, anderzijds zijn veel criteria voor het vaststellen van effecten op soorten en ecosystemen vanuit een ecologisch perspectief te beperkt. Dit geldt vooral voor de selectie van beschermde soorten en juridische beschermingsformules. Hierdoor blijven belangrijke potentiële effecten op biodiversiteit en beschermde soorten in juridische procedures buiten beschouwing. Definities van veel ecologische termen in de wetgeving zijn soms onduidelijk en wijken vaak af van in de ecologische wetenschap gebruikte definities. In veel gevallen zijn de verschillen echter onvermijdelijk. De wetgeving dient ervoor te zorgen dat er een handhaafbaar stelsel van juridische normen bestaat, dat rechtszekerheid biedt en dat intern logisch consistent is. Implementatie van alle ecologische inzichten, die zich bovendien steeds blijven ontwikkelen, zal in veel gevallen conflicteren met bovengenoemde criteria. Er kan echter wel worden gezocht naar mogelijkheden om de verschillen te overbruggen.

Om de wetgeving doelmatiger te maken voor natuurbescherming en ecologisch onderzoek relevanter voor juridische procedures, moet meer aandacht worden gegeven aan de ecologische betekenis en de kans op uitsterven van soorten en ecosystemen. Meer aandacht is nodig voor ecosysteemdynamiek (waaronder sleutelprocessen zoals de invloed van micro-organismen op de chemie van water en bodem, omgevingsdynamiek, successie en uitwisseling van genen), ecologische schaalniveaus, ruimtelijke heterogeniteit en de structuur en functie van ecologische netwerken. Bovendien kan in de wetgeving beter worden omgegaan met onzekerheden in effectenbeoordeling. Concrete mogelijkheden voor het overbruggen van de verschillen tussen ecologie en wetgeving zijn het opstellen van ecologisch goed onderbouwde beheerplannen voor beschermde gebieden en gedragscodes voor vrijstelling van de Flora- en Faunawet. Ook jurisprudentie vult vaak gaten in de wetgeving op en biedt dus ook concrete mogelijkheden voor verbetering van de inbreng van ecologische kennis en informatie in juridische procedures. Tenslotte kan worden gedacht aan het opstellen van kwaliteitsnormen voor ecologische effectbeoordelingen. Ecologische effectbeoordelingen dienen terdege aandacht te geven aan cumulatieve effecten en kunnen vaak het best gedaan worden op het niveau van het regionale metapopulatienetwerk. Op ecologisch vlak is met name meer inzicht nodig in verspreiding van beschermde soorten, hun habitat, ecologische netwerken en de respons van soorten en ecosystemen op menselijke activiteiten. Tenslotte zou meer consensus over ecologische relevantie van soorten en definities van ecologische termen zeer waardevol zijn.

De juristen en bestuurders die de huidige natuurbeschermingswetgeving toepassen zouden moeten proberen meer te begrijpen van een ecologische benadering voor natuurbescherming. Ecologen dienen zich te realiseren dat de wetgeving een denkwereld op zichzelf vertegenwoordigt, met haar eigen doelen (bijvoorbeeld het bieden van een ethisch kader voor omgang met de natuur, rechtszekerheid en handhaafbaarheid) en criteria voor waarheidsclaims (zoals logische consistentie).

7 Synthese en conclusies

In de synthese (hoofdstuk VII) worden de mogelijkheden en beperkingen van BIO-SAFE samengevat en aanbevelingen gedaan voor verbetering van het model. Ook wordt ingegaan op de implicaties van natuurbeschermingswetgeving voor inrichting en beheer, de effecten van rivierbeheermaatregelen op beschermde en bedreigde soorten en mogelijkheden voor integratie van ecologische kennis met wettelijke instrumenten voor natuurbescherming in het rivierbeheer.

Op basis van de resultaten van hoofdstuk II t/m VI wordt geconcludeerd dat BIO-SAFE een operationeel en wetenschappelijk onderbouwd model voor integratie van ecologische kennis en informatie met wettelijke instrumenten voor natuurbescherming in het rivierbeheer is. Het levert bruikbare informatie voor evaluatiestudies, het is valide voor effectenbeoordeling en weinig gevoelig voor waardetoekenning. Belangrijke ecologische beperkingen van het model betreffen genetische en populatieaspecten (beide niet in het model verwerkt), beschrijving van habitats (slechts gebaseerd op ecotopen) en effectvoorspelling (die alleen is gebaseerd op de relatieve oppervlakte van ecotopen en soort-ecotoop koppeling). Er zijn ook juridische beperkingen van het model. BIO-SAFE is niet in alle onderdelen van juridische procedures bruikbaar, betreft alleen rivierkarakteristieke soorten en beoordeelt effecten niet op het niveau van afzonderlijke soorten. Andere juridische beperkingen betreffen de waardetoekenning, die niet altijd correspondeert met het juridische kader.

De resultaten van de modeltoepassing in hoofdstuk II t/m V betreffen het belang van ecotopen, de effecten van in uiterwaarden genomen maatregelen en beoordeling van te verwachten effecten van herinrichtingsplannen. Ecotoopwaardering laat zien dat elke taxonomische groep aan verschillende sets van ecotopen is gerelateerd en dat geen ecotopen en delen van de hydrodynamische gradiënt kunnen worden genegeerd. Nu in het rivierbeheer het accent ligt op het verhogen van dynamiek, moet hier krachtig worden opgemerkt dat ook middelmatig en laagdynamische ecotopen belangrijk zijn. Maatregelen voor ecologisch herstel en herinrichting van uiterwaarden hebben zowel positieve als negatieve gevolgen voor beschermde en bedreigde rivierkarakteristieke soorten. Over het algemeen zijn de positieve effecten veelbelovend maar beperkt. Beoordeling van effecten vooraf toont aan dat de meest natuurlijke situatie zoals planvormers die voor zich zien niet altijd correspondeert met de hoogste potenties voor beschermde en bedreigde rivierkarakteristieke soorten. Plannen die een toename van rivierdynamiek in uiterwaarden betekenen zullen in sommige gevallen leiden tot negatieve of suboptimale effecten. Maatregelen kunnen worden geoptimaliseerd door het vergroten van de ruimtelijke schaal en het gebruiken van gedetailleerde kennis over de potenties van een bepaald gebied. Als de landschapsheterogeniteit wordt gemaximaliseerd, en middelmatig en laagdynamische omstandigheden worden behouden en ontwikkeld, kan natuurbehoud worden gecombineerd met veiligheid en ecologisch rehabilitatie. Dit vereist dat ook loodrecht op de rivier naar ruimte wordt gezocht, en er creatief moet worden omgegaan met de beschikbare ruimte.

Het ecologische en het juridische kader voor natuurbescherming hebben verschillende doelen, criteria voor het maken van waarheidsclaims, benaderingen voor natuur en het selecteren van soorten en begrippenkaders. Er zijn echter verschillende mogelijkheden voor het overbruggen van deze verschillen in de praktijk, bijvoorbeeld in beheerplannen, gedragscodes, effectenstudies en jurisprudentie. BIO-SAFE draagt bij aan integratie in het rivierbeheer door het combineren en wederzijds aanpassen van ecologische kennis en wettelijke instrumenten in een model dat bruikbare informatie levert aan de rivierbeheerder.

BIO-SAFE kan worden geoptimaliseerd in ecologische zin door het gebruiken van meer gedetailleerde ecotoopclassificaties, het bepalen van minimaal benodigde oppervlakten voor soorten en populaties en algoritmen die non-lineaire relaties tussen ecotoopoppervlakte en potenties voor soorten en taxonomische groepen kunnen beschrijven. Het verbeteren van het model in juridische zin zou de mogelijkheid moeten bieden effecten te beoordelen op het niveau van afzonderlijke soorten, en rekening te houden met de complexiteit van de juridische status van soorten. BIO-SAFE kan het beste vroeg in planvormingsprocessen worden gebruikt voor het ontwerpen van inrichtingsscenario's die de mogelijkheden van beschermde en bedreigde soorten optimaliseren, tijdens de bestekfase om te bepalen welke ecotopen in de actuele situatie het meest waardevol zijn en daarom beschermd moeten worden, en als een instrument voor evaluatie van in uiterwaarden genomen maatregelen.

Integratie in het rivierbeheer in brede zin betekent dat meer aandacht nodig is voor sociaal-culturele en psychologische aspecten van riviersystemen, en dan vooral aan visuele aantrekkelijkheid, identiteit en cultuurhistorie van het rivierlandschap. Percepties van burgers en lokale overheden spelen hierbij een belangrijke rol. Bovendien moet rekening worden gehouden met de supranationale dimensie van rivierbeheer, de interactie tussen rivierbeheerders en het systeem dat ze besturen, en de interactie tussen rivierbeheerders en de burgers die daar leven. Integratie en interactie in het rivierbeheer zijn grote uitdagingen voor de toekomst.

Dankwoord

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Curriculum vitae

Reinier de Nooij werd geboren op 4 Augustus 1976 in een plaats waar alle sociale, ecologische en economische extremen te vinden zijn: Mexico-stad. Nadat hij op 2-jarige leeftijd met zijn ouders Johan en Immy terugkeerde naar Nederland, kreeg Reinier gezelschap van zijn jongere broer Floris. Hij volgde een VWO-opleiding aan het Titus Brandsma Lyceum te Oss, waarna hij Natuurwetenschappelijke Milieukunde, met als tweejarige basisdiscipline Biologie, studeerde aan de Radboud Universiteit Nijmegen (RU). Na zijn afstuderen in 1999 ging hij als junior-onderzoeker werken bij de afdeling Milieukunde. Uit dit onderzoeksproject vloeide later zijn promotieonderzoek voort. In 2004 trouwde hij met Mirjam, de liefde van zijn leven, op de dag dat zij 10 jaar samen waren.

Al als kind en puber was Reinier zeer betrokken bij planten, dieren en milieuproblemen. Hoewel hij op de middelbare school uitblonk in talen, geschiedenis en levensbeschouwelijke vakken, trokken de biologie en de milieukunde Reinier toch het sterkst aan. Vandaar dat hij in Nijmegen ging studeren: hier was het mogelijk om eerst twee jaar biologie te doen en vervolgens af te studeren in de milieukunde.

Zijn onderzoeksstages betroffen natuurherstel, natuurvriendelijk rivierbeheer en planstudie voor herinrichting van uiterwaardgebieden. De eerste stage was een onderzoek naar herstelmaatregelen voor een nat elzenbroekbos in Limburg, dat werd uitgevoerd bij de afdeling Aquatische Ecologie en Milieubiologie (RU). Hierna liep Reinier stage bij de afdeling Milieukunde van de RU, waar hij een beoordelingsmethodiek voor plannen voor natuurvriendelijke oevers van rivieren ontwierp en toepaste. Tijdens dit onderzoek werd zijn belangstelling gewekt voor de activiteiten van Rijkswaterstaat. Daarom ging hij begin 1999 bij deze organisatie stage lopen. Bij de Directie Oost-Nederland werkte hij mee aan het ontwikkelen van een herinrichtingsplan en milieu-aspectenstudie voor de Rosandepolder, gelegen langs de Nederrijn aan de voet van de stuwal onder Oosterbeek.

Toen Reinier eind 1999 met een onderzoeksidee aanklopte bij de afdeling Milieukunde bleek men daar al vergevorderde ideeën te hebben voor een tweejarig onderzoeksproject genaamd ‘BIO-SAFE’ dat werd gefinancierd door de EU. Reinier werd uitgenodigd te solliciteren, en vervolgens aangenomen. In 2000 en 2001 ontwikkelde hij een transnationale versie van het model BIO-SAFE en bouwde het model verder uit zodat er ook scenariostudies mee gedaan konden worden. Toen bleek dat er mogelijkheden waren voor een vervolgonderzoek dat kon uitmonden in een promotie, greep Reinier deze kans en diende een projectvoorstel in.

Vanaf 2002 werkte hij in deeltijd aan diverse onderzoeken die later de bouwstenen voor zijn proefschrift zouden vormen. Daarnaast was hij jarenlang betrokken bij een kunstproject genaamd ‘de Faculteit der Medemenselijkheid, Persoonlijke Groei en Duurzame Ontwikkeling’.Voor dit kunstproject, dat mede werd georganiseerd door het Universitair Centrum voor Milieuwetenschappen en Duurzame Ontwikkeling (UCM-DO), ontwikkelde en gaf hij diverse workshops, leidde debatten en nam deel aan artistieke manifestaties. Na het einde van zijn dienstverband bij de afdeling Milieukunde werkte hij enige tijd bij het UCM-DO. Dit combineerde Reinier met het schrijven aan zijn proefschrift.

Reinier is in toenemende mate geïnteresseerd geraakt in natuurbeschermingswetgeving en de relatie tussen wetgeving en ecologisch onderzoek. Hij wil in de toekomst een bijdrage leveren aan het versterken van de samenhang van deze twee vakgebieden en aan verbetering van de toepassing van de wet- en regelgeving voor natuurbescherming. Een ander belangrijk interessegebied is integratie van natuurwetenschappelijke, sociale, culturele en psychologische kennis in processen in het rivierbeheer.

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