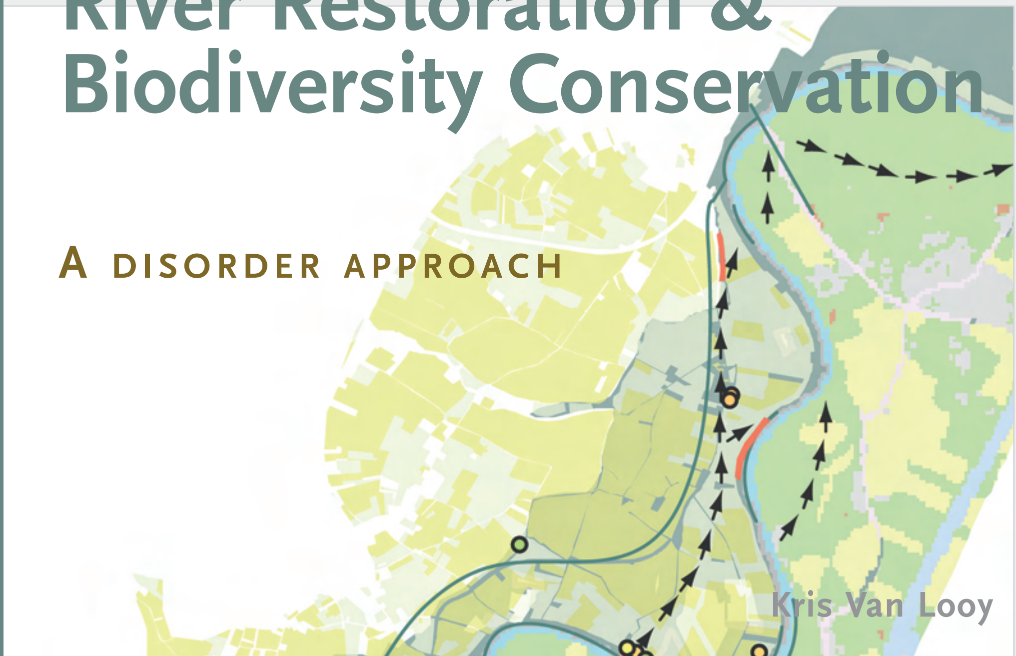
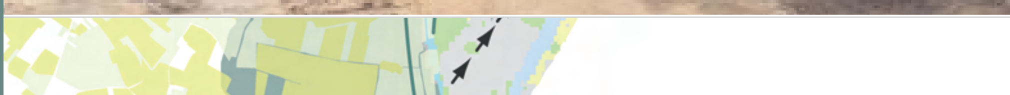
River Restoration & Biodiversity Conservation

A disorder approach



Kris Van Looy



Universiteit Antwerpen

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Kris Van Looy

Proefschrift voorgedragen tot het bekomen van de graad van Doctor in de Wetenschappen

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Acknowledgements

I want to thank all the colleagues and friends that shared thoughts, visions, ideas and joy in river restoration with me and that brought me this far. The River Meuse inspired me in this work, gave me valuable friends and brought me in many nice places, so maybe I owe her some thanks too.

Special thanks goes in the first place to Geert De Blust for the enlightenment, there was always some enlightened ideas at hand right next door. Olivier Honnay for talking me into this late-night job and all the support. Of course Patrick Meire who taught me a lot, for the encouragement also and not the least for the competition between Flanders' streams, of which I had the honour to study the largest and most natural, but that discussion remains, I guess and hope. Eckhart Kuijken for the trust and the persistence to harden me.

The river authority AWZ granted this research for a good part, was very cooperative in the experiments and the experiencing in the Meuse restoration efforts. The enthusiasm of Herman Gielen is not mentioned as important driver for restoration and conservation in this thesis, but it definitely is.

And last but not least, Anouck, Siemen and Wannes, who make it all worth it.

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Introduction, aims and outline

1. Introduction

In two consecutive winters of the mid-nineties of the last century, the highest ever recorded peak discharges shook-up the Meuse valley and its inhabitants. A catastrophe to man, a blessing to nature; these extreme peak events revitalised the Common Meuse. Embankments were broken up, restoring the morphological processes of the river bed over extensive stretches. In the floodplain erosion and sedimentation processes reshaped the riparian landscape that was at the same time re-colonised by a range of riverine organisms, from aquatic macroinvertebrates to plant and bird species. This revitalisation was also source of inspiration for the restoration plans and research efforts, and it worked through in the appraisal and weight for the restoration programme that was initiated some years before. These events were the starting point of our research for restoration potentials in the Flemish Meuse valley.

River restoration and the natural value of river ecosystems is an increasing area of interest for management as well as research. There is an unprecedented need to preserve and restore aquatic and riparian biological diversity before extinction eliminates the opportunity (Kauffman et al. 1997).
The relationship between conservation and restoration can be one of excluding or complementing each other. Here we explore the latter option, and try to determine the merits of the restoration for the biodiversity conservation.
We can point at the pan-European legislative contexts and frameworks for both; the Habitat Directive aiming at the conservation of habitats and species and the Water Framework Directive oriented to the restoration of a good ecological status.

In this introduction, we will review the approaches and their conceptual frameworks, and further introduce the case-study of the River Meuse. Different headers are the conceptual frameworks, the biodiversity approach, river disorder and the Meuse case, with its general characterisation, mapping and sampling and restoration potentials and targets. At the end, an outline to this thesis is given.

1.1 Conceptual framework of conservation and restoration in river systems

Conservation and restoration

Conservation is both preservation and care, a dynamic process (Haslam 1996). Restoration is returning the system to a close approximation of the pre-disturbance ecosystem that is persistent and self-sustaining, though dynamic in its composition and functioning (Maurizi & Poillon 1992).
Haslam pointed at the dynamics in both definitions and the 'Sense of Place' in his plea for preliminary thoughts and theoretical considerations, before trying to convert rivers to a 'Standard Recommended' river without sense of the unique character and special features of the river.

In this context it is important to refer to a clear definition of conservation and restoration options and dimensions. At one extreme, conservation goals reflect the desire to preserve remnants of natural or near-intact systems. Far more common, however, are endeavours to rectify and repair some (or all) of the damage to river ecosystems brought about by human activities. Various terms used to describe these goals and activities can be summarized using the umbrella term "restoration".

Boon (1992) describes five appropriate strategies for river conservation or "restoration", in accordance with the state of the river. Where few natural or semi-natural systems with untouched hydrodynamics remain, their preservation is the task. This is rare in Europe, where all large rivers are more or less controlled. For rivers with a still high ecosystem quality and with ecological key factors functioning without major impediments, there the management option is for limitation of catchment development. When the quality is low, their mitigation becomes the case and the development of existing economic and recreational functions need to be accompanied by the implementation of measures that allow the survival of habitats and organisms. When rivers are degraded to a point that natural hydrodynamics are hardly recognisable and only scattered and small remnants of populations persist, there the emphasis shifts towards river restoration. With the help of well chosen restoration techniques and nature development projects, more suitable habitats need to be created, enhancing the recovery of the remaining populations and the establishment of new ones (Gore 1985). The final management option mentioned by Boon (1992), is for the worst case scenario where recovery is hopeless and dereliction is the only wise decision. In these cases, limited resources should not be allocated, but rather directed towards more promising restoration projects. As Boon gives 5 definitions of restoration 'dimensions', we try to see the continuum in these terms and frequent the most commonly used term river restoration, often without regard to a historic reference, but in the broad sense of restoring/enhancing river functioning and specific river communities. In the continuum of restoration types, targets range from strong ecological goals to hard engineering erosion control and containment efforts (Giller 2005). Emphasis in our work is on the habitat restoration and biodiversity conservation scope of river restoration projects.

These observations bring us back to the definition of Maurizi & Poillon, for which the second part is essential and points at the central aspects in this thesis, the dynamics in composition and functioning as key features in the river ecosystem.

Dynamics and equilibrium

The disturbance regime is unquestionably one of the most characteristic aspects of the river system. Yet, from a managerial viewpoint, same for river manager as for nature conservationist, these dynamics are a stand-in-the-way for a concise management and planning.
Being humans, we try to identify equilibrium conditions in this spatial and temporal heterogeneity. This equilibrium can then be a desired 'controlled situation' in stability terms;

- for nature: good ecological status, favourable conservation status, consisting of
    - viable populations of target species
    - sustainable habitat networks
    - good habitat quality in environmental conditions
- from river manager's point of view: stability and safety
    - protection against flooding
    - water supply

Nevertheless, for both exigencies, a certain amount of freedom is to be allowed to the fluvial dynamics in order to sustain its biodiversity as well as stability. The resulting conditions description should be for a dynamic equilibrium.

The river's dynamic equilibrium concept

Figure 1 shows the equilibrium between the sediment alluvial flow (Qs) and the water flow (Q), which provides, with the slope, the energy able to carry the alluvial sediment. In a very simplified way, the fluvial dynamics is like a permanent oscillation of the pointer of a scale one of its pan is filled with sediment (variable Qs) and the other with water (variable Q). As these two elements are very variable in space and time, there is a permanent adjustment of the river morphology to erosion/sedimentation phenomena.

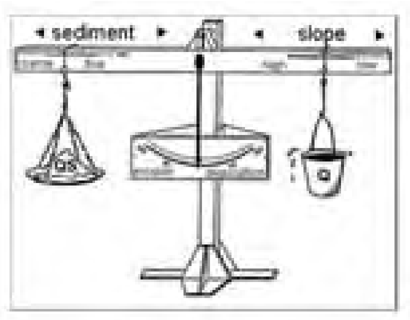


Figure 1.1 : The Lane's scale (1955) showing the dynamic equilibrium concept

It is a dynamic equilibrium. But if, for example, the sediment transport decreases for a long time (dams, gravel pits,...), the river erodes the bottom of its bed or its banks, in order to fill up again its bedload transport capacity. If the banks are protected (vegetation, rockfill,...), the phenomenon of re-equilibrium takes place only on the bottom of the bed (incision) and new equilibrium conditions evolve over the whole of the river's functions and structures.

Reversibility and irreversibility

Leopold (1964) defined the hydraulic geometry as a specific equilibrium condition; a most probable form to which the river progresses, searching equilibrium in the two somewhat opposing processes of maximizing dissipation of energy in the tendency to minimum work, and the tendency of equal distribution of the power expenditure. So, every state of the system is a balanced weighing of these opposing tendencies with an inherent variability.
These tendencies result in specific physical operations and changes in the river channel, like successive local acts of erosion or deposition , dictated by available forces exerted by the flowing water and countered by resisting forces of bed, bank and transported load.
For a river system in an equilibrium stage, the question is for the definition of irreversible processes, responsible for changes in function and structure. Changes in landuse in the catchment, regulation activities, at what point are they responsible for structural and irreversible changes in the river system? Only when specific nonlinear stochastic equations are altered, when specific conditions of bed geometry are altered giving way to a new entropy regime (slope/substrate).
In theory, a pressure is reversible when, in case it is omitted, the system returns to its natural equilibrium state and ecological integrity, as defined in the reference condition. Because this definition is hard to handle, a more pragmatic approach is proposed (i.e. in the light of the management plans for the Water Framework Directive). Alterations are determined as irreversible if caused by general changes in the land use of the catchment area or intrinsic to functions like shipping, urbanisation with no perspective of cessation/termination of these functions in the near future.

A good prediction of reversibility is necessary to assess to what extent rehabilitation is possible. Therefore there is need for indicators and geomorphic criteria for river systems in general and restoration programmes. For large rivers, which are in general mostly heavily modified, the identification of good indicators of pressure-impacts is crucial in the assumptions and evaluation of river rehabilitation (Lorenz et al. 1997).

The definition of reference conditions is in this respect also essential, and a specific demand in the WFD, for the definition of the high ecological quality status. The use of references in understanding the complexity of the natural riverine landscape is essential for assessing the extent to which riverine ecosystems have been altered by human activities and for designing and carrying out effective managerial strategies (Ward et al. 1999). Lack of this knowledge makes us underestimate the role river dynamics play in sustaining the ecological integrity. Therefore references play a key role in the definition of indices and measures for specific river landscape features, useful as tools in the design of restoration plans.

Biological strategies of resilience and disturbance

Resilience is the natural capacity to recover from an alteration or the adaptation to a regular disturbance. Promoting the resilience in the ecosystem means strengthening the local communities and mostly conserving the actual conditions.
Disturbance is the perturbation of the system by an irregularity in physical conditions, disturbance strategies take profit of these irregular processes. So, promoting the disturbance strategy means especially promoting and restoring 'natural' dynamics disregarding/notwithstanding local/present conditions. This can of course promote resilience in the long run too.

Organisms and communities in the river system show adaptations allowing them to react to changes in the environment. Biological strategies of disturbance and resilience in the river system can be distinguished to derive guidelines for a self-sustaining healthy river ecosystem.
We focus on disturbance and resilience aspects in ecosystem/biodiversity responses to pressures/restoration measures. Quantitative measures are derived from quantified relationships for ecosystem processes and between pressures-impacts and species/communities. These relations and measures are the cornerstones in the defining and designing of restoration potentials and options.
We followed the biodiversity approach in these analyses of key ecological factors for the River Meuse. For this approach, emphasis is on the richness in characteristic communities and species in the scope of the entire river ecosystem sustaining healthy biocenoses.

1.2 Biodiversity approach

The concept of biodiversity represents a broad integrative perspective of diversity in the natural system, with a strong concern for threats to gene pools, species and habitats on a global scale (Ricklefs & Schluter 1993). Biodiversity conservation involves more than just species diversity or endangered species, and asks for a scale-sensitive and hierarchical approach (Noss 1990). Noss distinguishes four levels of organisation in a hierarchical framework of biodiversity: the landscape level, the community/ecosystem level, the population/species level and the genetic level.
Understanding the factors that structure diversity patterns of local species assemblages requires knowledge of processes that determine species richness at the regional level and the rates of spatial turnover of species (Caley & Schluter 1997). Gamma diversity, the total number of species in a region, is a function of the number of species per habitat (alpha diversity), the number of habitats (habitat diversity), and the turnover of species between habitats (beta diversity) (Ward et al. 1999).
According to Ward et al. (1999), The spatio-temporal heterogeneity of riverine landscapes, responsible for its biodiversity, can be attributed to three determining components: ecotone properties, ecological connectivity and successional processes.

- The riparian landscape consists of a transition zone forming a complex gradient between the river channel and the uplands or terraces, structuring species composition and richness patterns (Naiman et al. 1988) and resulting in a high habitat heterogeneity.
- The river corridor is generally acknowledged for its strong connectivity. Still it remains important to distinguish between processes and agents, as connectivity also refers to the extent to which water, nutrients and organic matter cross the riparian landscape. The hydrological connectivity is well-known for structuring biodiversity patterns (Heiler et al. 1998; Ward et al. 1999).
- Successional processes of the riparian landscape include flooding and channel migration as larger-scale spatially and temporally structuring processes. Succession in strict biotic sense is less addressed in the riverine landscape and biodiversity patterns, yet it also plays a strong role at community and species turnover level.

The succession of vegetation in riparian zones is hardly addressed, as processes of vegetation dynamics in the riparian landscape are mostly immediately linked to hydroregime and one to one physical relations. Not only primary succession, but also many secondary and complex successional patterns govern the riparian area, with interactions of many physical processes of inundation, drought, sedimentation, shear stress, herbivore and plant adaptations and strategies.
Result is a complex landscape with disturbance regulation of local potentials for dominance or conditioning the invasion by additional species.

To understand the biodiversity of river ecosystems, it is necessary to appreciate the central organizing role played by a dynamically varying physical environment (Poff et al. 1997). In rivers, the physical structure of habitat is defined largely by the movement of water and sediment within the channel and between the channel and floodplain. Reduction of ecological heterogeneity reduces options for species diversity (Naveh & Lieberman 1994). Ecological heterogeneity in river systems is closely related to flow regime and flood pulse characteristics, influenced by river management and floodplain land use.
Biodiversity approaches start mostly from a certain reference that is guided by riverine communities and species, offering quantifiable responses to abiotic conditions and anthropogenic pressures. Therefore this approach offers good perspectives in the scope of defining key processes and targets in a tangible manner.

Biodiversity analysis

For its analysis, biodiversity consists of diversity in species richness and in composition.

For species richness, the species-area relationship is an essential feature and its assumed linear relationship the basis for the island theory of Macarthur & Wilson (1967), a key concept in ecology. A dynamic equilibrium between colonization and extinction, the one compensating for the other, in geographical units is believed to lie at the basis of this theory.
Recent research puts more emphasis on heterogeneity at the basis of biodiversity patterns, and a gradient is drawn between the deterministic island theory equilibrium, and the stochastic non-equilibrium conditions, present in extremely heteroge-

neous environments (Wiens 1984).
Community composition and diversity is documented to react to heterogeneity both caused by physical conditions and by specific plant life-trait combinations (Hérault & Honnay 2005). Discontinuities in community composition and diversity patterns can be related both to environmental factors as to recruitment and dispersal limitations of species (and the interplay of both).

The number of species has low information content on the functional level of communities and ecosystems, as it does not necessarily respond to environmental changes. Measures for the relative abundances of species, like Simpson's and Shannon-Wiener's indices are the most commonly used. They can be used for single sites/samples, measuring alpha diversity (Whittaker, 1972), for more sites/samples determining the similarity as beta diversity and over more sample sets, called gamma diversity.
Where the Simpson's index is sensitive to dominance aspects, the Shannon-Wiener index has a strong indicative value for the heterogeneity in composition. This index belongs to the group of indices in the information theory. The array of indices in the group, measures the different aspects of information (order) or entropy (disorder) in the system (Orlóci et al. 2002). This disorder definition bridges the entropy interpretation between the physical and biological system and introduces the conceptual term upon which we will build our conclusions.

Biodiversity analysis at different scale levels

By means of a proportional analysis of different information/entropy indices, the interpretation of ecosystem species diversity can highlight crucial components in structure and richness of communities. Different entropy functions allow the identification of processes at the root of richness and composition aspects (Orlóci et al. 2002). This biodiversity analysis of different richness and structure components is a useful approach when dealing with a range of scale processes (Pollock et al. 1998). Linking diversity in species numbers and life-forms to environmental gradients and general landscape features, can be a guide in the development of conservation and restoration strategies (Wang et al. 2002).

For our data of floodplain meadow vegetation for the different reaches along the River Meuse (see II.2), we can analyse these biodiversity measures. The entropy functions and proportional analysis are measured following Orlóci et al.(2002). The first order entropy function (Shannon index) is the most currently used measure of diversity. The use of the different order entropy functions can give further information on diversity in composition and richness of species; the zero order entropy is an index of species richness, the second order entropy of similarity. The first order entropy is defined as the level of disorder in the data over reaches.
The order zero entropy index of species richness (Table 4, Fig. 2), gives an upper limit of entropy in the species data matrix. The entropy remains high over the first and second order in the structure of the meadow data.

Table 1.1 Diversity components in entropy values and their shares.

Diversity components		Entropy order		Maximum (2,5849)	
component	element			% of total	% of max
Richness	species	H0	1.87	42	72.3
Disorder	structure	H1	1.71	38	66.2
Similarity	evenness	H2	0.85	19	35.9

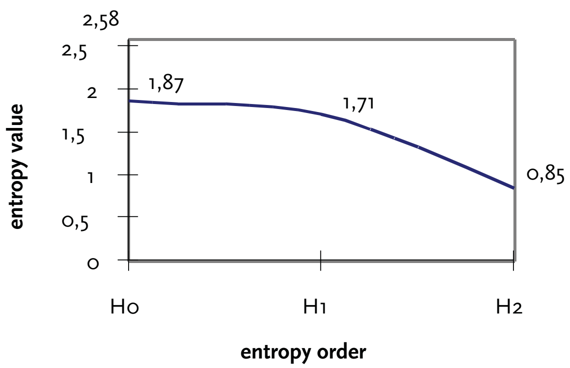


Figure 1.2 The entropy function shows the different levels of diversity and heterogeneity in the floodplain meadow communities over the reaches.

The entropy function decreases slowly with a very high order one entropy (Shannon index), compared to the maximum and the total. The disorder is not only present in the overall dataset of the floodplain meadows in the river basin, but also at the reach scale. So, the diversity in composition and species richness of the floodplain meadow vegetation is high between and within the reaches. The disorder in our dataset is high compared to other studies (e.g. Ward 1998; Deiller et al. 2001; Orloci et al. 2002; Wang et al. 2002) and remains present through the entropy levels. Put simply, disorder determines community structures at river basin scale as well as at the local patch scale. This observation also confirms the conclusions of Pollock et al. (1998) that river heterogeneity works at different scale levels.

Hérault & Honnay documented heterogeneity in communities due to different responses of life traits, in their case for regeneration and history traits. We detected a heterogeneous response in groups of structure traits to one of the main drivers of biodiversity in the river system, the flood frequency.

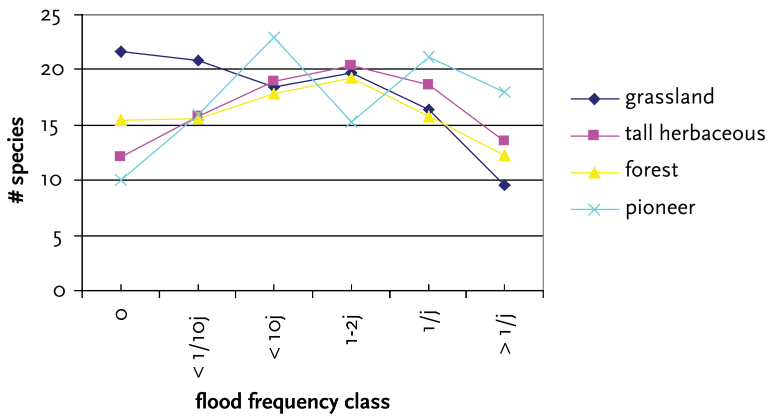


Figure 1.3 Average species richness for the different vegetation structure plots over five flood frequency classes. Grassland: 180 plots, pioneer:86 forest 105, tall herbaceous vegetation:94.

Different patterns for the life traits are immediately apparent and some conclusions to flooding sensitivity and intermediate disturbance aspects come forward, as will be discussed in the further analyses. The dry river grasslands are a remarkable asset of the Meuse valley, obvious for their richness in this diagram, same as the intermediate disturbance responses of forests and tall herbaceous communities. The diversity pattern of tall herbaceous vegetation and forests conforms to the Intermediate Disturbance Hypothesis (see III.1), while grassland and pioneer vegetations do not conform to this assumed general rule.

1.3 What is river disorder?

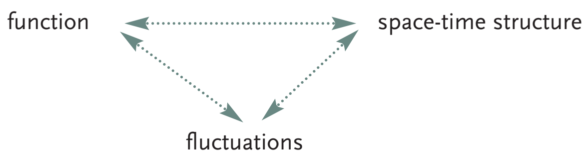
Disorder in the physical system

The concept of entropy in the river system

When we include the energetic and material fluxes in the river system as essential features in our description, and put the space-time interactions in the perspective of stochasticity, the concept of entropy is unavoidable and its derivations offer very useful observations in the field of our research.
Open systems like the river move toward a dynamic equilibrium. In this ideal steady state, parameters are kept constant in spite of matter and energy exchanges with the environment, and the conservation of structure is enabled by minimum entropy production and by maximum order or negentropy.

Prigogine (1973) defined aspects of biological order in terms of non-equilibrium thermodynamics. Thermodynamic non-equilibrium is not only a state of disturbance from equilibrium, but is also a source of order and organisation. He founded a non-linear thermodynamics of irreversible processes, enabling the description of the spontaneous formation of structures in open systems that exchange energy and matter with their environment and lead to the evolution of new, dynamic, globally stable systems. This "order through fluctuation" concept based on the definition of dissipative structures is of major importance in our conceptual approach. The non-equilibrium condition of the river system can be described in terms of dissipative structures maintaining continuous entropy production, which does not accumulate in the system, but is part of the continuous energy exchange with the environment. This exchange of matter and energy is the source of this inner non-equilibrium with important exchange processes. With this "coherent" system behaviour, order is maintained beyond instability thresholds. If fluctuations exceed a critical size and no longer can be absorbed by energy exchange, these structures are driven beyond a threshold to a new regime, and thus a qualitative change in the system's dynamics is introduced. A new regime of entropy production and organisation and order will be installed. This is the principle of order through fluctuation. The dissipative structures can be considered as giant fluctuations, leading to a whole spectrum of characteristic dimensions in functioning and structures of the system.

The level of description of a system can be represented as follows (Landscape ecology, Naveh & Lieberman 1984):



Prigogine doesn't only open the world of near-equilibrium conditions and the higher order organisation of non-equilibrium conditions. He points at disorder aspects as basic elements determining the equilibrium state to develop (Prigogine & Stengers 1993). So it's not the order elements that are the triggers and that we have to focus in describing a systems functions and structure, but the disorder aspects.

Disorder in the biotic system

The physical concept of disorder in the field of entropy and thermodynamics, is also applicable in the biotic system. Especially considering aspects of diversity, the entropy values and disorder measures prove useful concepts (Orloci et al. 2002).
River systems belong to the most species rich and divers ecosystems of the world's temperate regions. This is mostly considered to be the result of local environmental heterogeneity and the complexity of river processes, allowing for many species with a variety of life traits and population strategies to coexist.

In this perspective, intermediate levels of disturbance are observed to generate maximum diversity (Intermediate Disturbance model, Connell 1978), or in combination with productivity the highest diversity is observed at intermediate disturbance and productivity levels (Dynamic Equilibrium Model, Huston 1979). This model predicts that the potential number of competing species in a habitat varies according to the relationship between disturbance and productivity (resources). At high levels of disturbance, biodiversity is maximized in habitats with abundant resources (translated in a high rate of population growth), whereas at low levels of disturbance, maximum diversity is attained in habitats with sparse resources.
In general, riparian plant communities are composed of specialized and disturbance-adapted species within a matrix of less-specialized and less-frequently disturbed upland community species (Naiman & Décamps 1997; Henry et al. 1996). Mostly, river system descriptions and river quality assessments focus mainly on the strict riverine communities and species. Characteristic river species with specific adaptations are selected as key species for protection of river ecosystems.

The described extreme habitat heterogeneity and the consequent hyperdiversity of the vegetation in the river corridor are questioned for their contribution to biodiversity conservation, as the floodplain might function as sink habitat for a major part of the present species (Mouw & Alaback 2002). These authors argue that only specialist species of river corridors are of conservation interest, as the other species might not produce stable, enduring populations in the river corridor.

We describe the river as an open system, with the disorder elements contributing to the entropy patterns of material, organisms and energy dissipation and disturbances in connectivity along the corridor and between landscapes. River networks dissect landscapes and provide a natural framework for conservation planning, with distinct additive value to biodiversity conservation, if these indeed influence diversity patterns (Sabo et al. 2005). In our observed patterns for the River Meuse, we found prove for the influence of the dissected and connected landscapes in the river network.

The described theories and appreciation of river ecosystem functions in relation to disturbance work only within the borders of the local scale patterns and processes, as the described biodiversity patterns are influenced by small-scale patterns of spa-

tial heterogeneity (Pollock et al. 1998). At regional scales and integrating chorological factors at landscape level, important factors of interactions between populations by species dispersal come at issue/to light. Colonisation and extinction of local populations are important aspects of community composition and diversity over larger spatial and temporal scales, for the definition of conservation strategies (Hanski 1999).
The spatial distribution of species is proposed as a structural ecosystem indicator since it is an aggregation of underlying functional processes (Lorenz et al. 1997).

Order and disorder in the river continuum

The assumption of continuity, one of the key concepts in most frameworks for river ecology, was already argued by Statzner & Higler (1985). Although there's no discussion about equilibrium conditions between the biological communities and the physical environment, the continuity in changes along the river, as stated in the River Continuum Concept (Vannote et al. 1980), is not a continuous process with simple relation to river order. Tabacchi et al. (1990) already described discontinuity in the longitudinal changes of riparian vegetation composition. Rapid changes were explained by transitions from higher to lower altitude and tributary influence. The longitudinal vegetation discontinuities they described did not generally coincide with environmental change. The here determined entropy disorder in the biotic communities shows conformity over the reaches with the physical variables of stream power, width of the floodplain, number of tributaries and adjacent ecoregions. The open system character of the river guarantees that these variables play a role at reach scale more than just locally near confluences or other discontinuities.
To this disorder character contribute in a variable, often complementary, way the connectivity aspects within the catchment and the physical disturbance regime of natural and anthropogenic perturbations. The dissipation of energy in the creation of habitat heterogeneity, together with the maximization of connectivity with contact to different ecoregions, results in changes in communities by influxes of species from the drainage network and the immediate surroundings and the loss of species that are less adapted to the changed environmental conditions. As this process of striving for equilibrium conditions in

regional communities by extinctions can only take place in combination with catastrophic events, or as Rosenzweig (1995) puts it 'accidents are needed for extinctions', the step towards non-equilibrium processes and 'disorder' is clear.

Where saturation and the striving for equilibrium are the foundations for community ecology, we observed patterns in assemblages with no or only secondary influence of resource limitation and competition (the processes behind these foundation patterns). More determining proved recruitment and dispersal limitation, extinction by fluvial or anthropogenic disturbance and responses to the changing physical conditions by resilience or disturbance strategies.
Fitter ea (1999) point at temporal variations leading to disequilibria at a point in space and to the possibility of coexistence of species which could not coexist if competition was allowed to proceed. We think most species assemblages in river systems can be categorized as non-equilibrium communities, not regarding whether it concerns aquatic macroinvertebrate assemblages, for which strong disorder is observed in the Meuse in frequent consecutive invasions of exotic species last decades (bij de Vaate et al. 2002), or riparian ground beetle assemblages responding to extreme local heterogeneity in flow dynamics (Van Looy ea 2005), or floodplain meadow vegetation where the inverse species-area relationship combined with the need for flood dynamics observed for the rare river corridor species, proves the opposing trend to saturation. So, these non-equilibrium communities are loosely structured assemblages with species more responding to environmental variations largely independent of one another (Wiens 1984). Especially the major contribution of stochastic events (extreme peak flows) to species dispersal and colonization/extinction, proves determining for observed diversity and composition patterns. As the persistence of small populations is strongly affected by stochastic problems (Foose ea 1995), our analysis of the population dynamics strategies of the rare river corridor species highlighted the threats for the characteristic river species of the Meuse and the conservation strategies for their survival and recovery.

Functional groups

Functional grouping is moreover introduced in predicting responses to changes in environmental conditions (Pillar 1999). Well-known examples are feeding guilds in

macroinvertebrates, used in description of relationships between biota and river size, geomorphological alterations and pollution; basis for ecosystem assessment methods (Bovee 1982; Karr 1999) In this grouping, disorder elements were identified for the River Rhine by Statzner & Higler (1985), feeding their comments to the RCC and the definition of the stream hydraulics concept underpinning the importance of hydraulic discontinuities for the structure and functioning of the biotic system.
For the rivers Rhine and Meuse, no balanced state was reached for macroinvertebrate communities during the process of ecological rehabilitation due to ungoing/consecutive invasions of non-indigenous species and alterations in habitat structure (Neumann 2002, bijde Vaate 2004). A stochastically changing relationship between water level fluctuations and functional habitat conditions concerning reproduction/competition/predation is considered main factor for this disorder.

In general, riparian plant communities are composed of specialized and disturbance-adapted species within a matrix of less-specialized and less-frequently disturbed upland community species (Naiman & Décamps 1997; Henry et al. 1996). The functional adaptations to disturbance include morphological and physiological as well as reproductive adaptations (Lavorel et al. 1997). Morphological adaptations include adventitious roots and root flexibility. Physiological adaptations include many metabolism adaptations to surviving flooding and anoxic conditions. Reproductive adaptations include trade-offs between sexual and asexual reproduction, seed size, timing of dormancy, timing and mechanism of seed dispersal (Hughes et al. 2000; Guilloy-Froget et al. 2002; Imbert & Lefèvre 2003). Looking at functional groups in the first place allows the distinction of the different factors at play. For a specific ecological unit or habitat patch these factors can be divided in local topological and regional chorological factors (Zonneveld 1995):

Local factors:

- soil humidity and productivity
- flood duration and timing
- management practice
- initial composition

Regional chorologic factors:

- flood pulse contact (~ isolation of floodplain areas by dikes)
- pattern of habitat at regional scale (~ fragmentation)
- stochasticity of high peak flows (~ disturbance by extreme peak flows)
- contact to uplands and adjacent landscape (~dispersal limitation)
- riparian corridor connectivity (~ recruitment limitation)

For the chorological factors the resulting processes and pressures that put species or communities at risk are added.
To detect the contribution of the disorder elements to composition and diversity of communities, we focussed on different functional groupings: 1. Life traits, 2. Population strategies and 3. Habitat templets

1. Life traits

For plant species, we can distinguish between regeneration traits and structural habitus and habitat traits; firstly pioneer, grassland, herbaceous or woody tall vegetations, and secondly xeric, freatophytic, calcareous, ... In our analysis we distinguished between pioneer/grassland/herbaceous-shrub/forest vegetations over the Common Meuse floodplain. When we select the xeric, calcareous river grasslands, we focus at the main protected habitat for the Common Meuse.

Dry calcareous river grasslands are present in the floodplain thanks to but also despite of the river dynamics. These communities are composed of a mixture of species from dry habitats all over the catchment, so for each river they show a strongly varying character in composition. Flood events are necessary to generate habitat for these communities by overbank sedimentation and for the dispersal of the plant propagules. Yet, these species are flood intolerant, not enduring longer inundations, wet soils nor nutrient enrichment. This illustrates the disorder character of the river system, accumulating elements and gathering its richness in a stochastic way from its location in the catchment. Regional factors are only recently acknowledged for their influence and merits to the composition and diversity of riverine communities (Hérault & Honnay 2005). This influence varies over functional groups with the life-traits considered. The merits of this contribution to biodiversity can be argued. Mouw & Alaback (2002) described this pattern as the hyperdiversity of riparian systems, with no trade off to biodiversity conservation. Sabo et al. (2005) give a broader view from a regional perspective of biodiversity conservation and the merits of a variety in species pools herein. When we regard the processes that deal with these regional factors, we think an important merit is present in this source/sink-functioning of river corridors.

With the influx of species from adjacent ecoregions these hyper diverse 'sinks' play a role of major importance to biodiversity conservation for populations can be viable for longer periods within the river system, as illustrated by the dry river grassland. This community of non-adapted species is given much attention, as it is taking benefit of the river's dynamic character and thus is a river-specific element, linked to the stochastic character and disorder of the river. River systems are recorded to play a key role in larger habitat networks and the remediation of fragmentation (Wilcox and Murphy 1985; Verboom et al. 2001; Sluis et al. 2001). The link between fragmentation and biodiversity and gene flow in the river corridor (Zwick 1992; Imbert & Lefèvre 2003; Van Looy et al. 2003) points at the crucial aspect of room for riparian ecosystems and their connectivity by fluvial processes.

2. Population strategies

Naiman & Décamps (1997) proposed a classification of plants into four broad categories of functional adaptations in population strategies: invader, endurer, resister and avoider, useful in understanding vegetation development and compositional aspects in river systems. Invaders produce large numbers of wind or water-disseminated propagules that colonize alluvial substrates. Endurers resprout after breakage or burial from floods. Resisters withstand periods of flooding during the growing season, while avoiders lack adaptations to specific disturbance types, so individuals that germinate in an unfavorable habitat will not survive. Endurers are well adapted to live under a number of disturbance regimes, while resisters are less broadly specialized river-adapted species, resisting specific stress conditions with a specific strategy. The above mentioned species of dry river grasslands are clearly avoiders, together with the invader species they were identified as related to the 'disorder' character in the catchment analysis. At the regional scale of reaches, landscape and site, all these adaptations give opportunities to react to disorder, as can be highlighted in the population dynamic strategies.

Freckleton & Watkinson (2002) defined population dynamic strategies explaining spatial dynamics of plants on a regional scale. They proposed a classification of large-scale spatial dynamics based on the relative importance of regional and local dynamics for the persistence of plant populations. The Freckleton & Watkinson typology provides a framework for the distinction of regional components of population dynamics, by integrating the key processes that determine the population dynamics (Eriksson, 1996; Hanski & Gilpin, 1997). It is a useful tool in determining how populations persist at the regional scale and important for the construction of conservation and rehabilitation strategies for species at risk (Freckleton & Watkinson, 2003, Jäkäläniemi et al., 2005). Population structure and spatial dynamics are recorded in many studies for their implications for conservation of riparian vegetation communities and endangered species (Van Treuren et al. 1993; Brys et al. 2003; Tero et al. 2003). The population dynamic strategies can be divided into resilience and disturbance strategies, referring to functional adaptations responsible for the spatial dynamics of populations. Disturbance strategies are characterized by high extinction-colonization and immigration-emigration ratios, as are present in shifting-cloud, source-sink and metapopulation strategies. Populations have resilience character when they show strong persistence in their occupied patches. This is the case for extended local, patchy and remnant populations.

The main distinction of the classification is between regional and local populations. In terms of the application of metapopulation theory, regional populations are relying on colonization from upstream populations. The species were assigned to one of these strategies without the evidence of a lengthy population study and no reference is made to current discussion on the distinction of metapopulations in non-continuous habitats and the evidence for extinctions and discrete habitat patch use (Gouyon et al. 1987; Ouborg 1993; Eriksson 1996; Freckleton & Watkinson 2003). Nevertheless this generalized strategy interpretation offers interesting opportunities to analyze aspects of species dispersal at a regional scale (Freckleton & Watkinson 2002).

The need for colonization from upstream (or outer river corridor) populations, is the disorder aspect in the river corridor. The detection of this dependence for species can be done by species populations monitoring over the reach, or by genetic analysis of present populations. We determined for Sisymbrium austriacum a strong invasive colonization pattern linked to extreme flood events, with further indications that the species needs these events for the development of sustainable populations in the region (Jacquemin et al. in Prep.). It is an in origin alpine species of the Pyrenees, accidentally introduced along the Meuse and nowadays widespread along the dynamic reaches with bars and overbank coarse sedimentation. As it is more widespread along the Meuse now than in its region of origin, it is called the Meuse rocket in our region.

3. Habitat templets

The habitat templet approach starts from a grouping based on habitat traits with a clear relation to species traits. Habitat templets for riparian ground beetles of the Meuse were derived from a clustering and ordination of species assemblages.
Dynamic riparian zones of gravel bars are habitat for species with adaptations allowing them to seek refuge under conditions of quick water level rises. These are wing development allowing them to flee and/or dorsal flattening allowing them to float. Under these dynamic bar conditions another group was detected of more sandy sediments, with species provided with digger front feat.
The disorder lies in the species' abilities for dispersal. Typical waterline dwellers were determined and observed to be quick colonisers of new habitat with abilities to bridge large distances quite easily. In a German research of riparian ground beetles with colour marks, one individual reappeared 800 km downstream! (Plachter et al. 1998). In our sampling, we also observed the remarkable presence of individuals of characteristic riverine species (i.e. Bembidion elongatum in 1998 and 1999 each year 1 individual was caught on a total of 15.000 individuals!) having only very restricted populations in the Meuse basin in some upstream tributaries (Lesse, Richir 2000)

Bringing this disorder to light is a hard task for science, as stochasticity and non or near-equilibrium conditions are a hard topic to investigate in an experimental setting and in a repeatable way.

Dispersal by flood events is evident and we described it moreover for the developing restoration sites (Van Looy & Kurstjens 1998, Van Looy 2005). Nevertheless it is hard to find evidence for the process itself, as the river has turbid waters at peak discharges, and not only buoyant seeds get dispersed. So, trapping seeds with nets only shows a partial aspect of floating propagules in the dispersal.
Evidence for dispersal and colonisation was also intriguing for the faunal communities in the system. In the ground beetle research experimental gravel bars were installed in the river bed to determine the contribution of active dispersal in colonisation and population dynamics and whether fragmentation of the riverbank habitat plays a role. This experimental setting was executed up to two times in 1999 and 2000, but got washed away in one week time at each attempt.
Finding evidence for the adaptations proves a hard task as well. For the Populus nigra research, young sprouts were transplanted together with mixtures of inbreeding poplar cultivars in an experimental setting. The complete setting at three gravel bars was washed out during the first winter.
Nevertheless we gathered some data to catch this disorder in facts and figures, and these present the core of our discourse.

1.4 A view of the River Meuse

Setting the scene of the River Meuse

The River Meuse is a large Northwest European stream. The Meuse has a rain-fed character, with a narrow upstream catchment, widening strongly in the middle reach with important subcatchments of Semois, Samber and Ourthe. Rocky primary soils in the large upstream Ardennes part, give quick runoff and strong flashy peaks. Modifications of flow regime and landcover in the catchment, endikements and land use intensification in the floodplain strongly perturbed the natural flow regime and the sedimentological and morphological processes in the basin. These changes go back to the first cultivation of the catchment area. River regulation goes back for millennia as the Meuse was already an important shipping way in Roman times.

Some reaches contain important natural values in the floodplain, important initiative is taken to protect the remaining habitat along the river and realize a chain of natural areas along the river to improve the river's corridor functioning by these stepping stones.

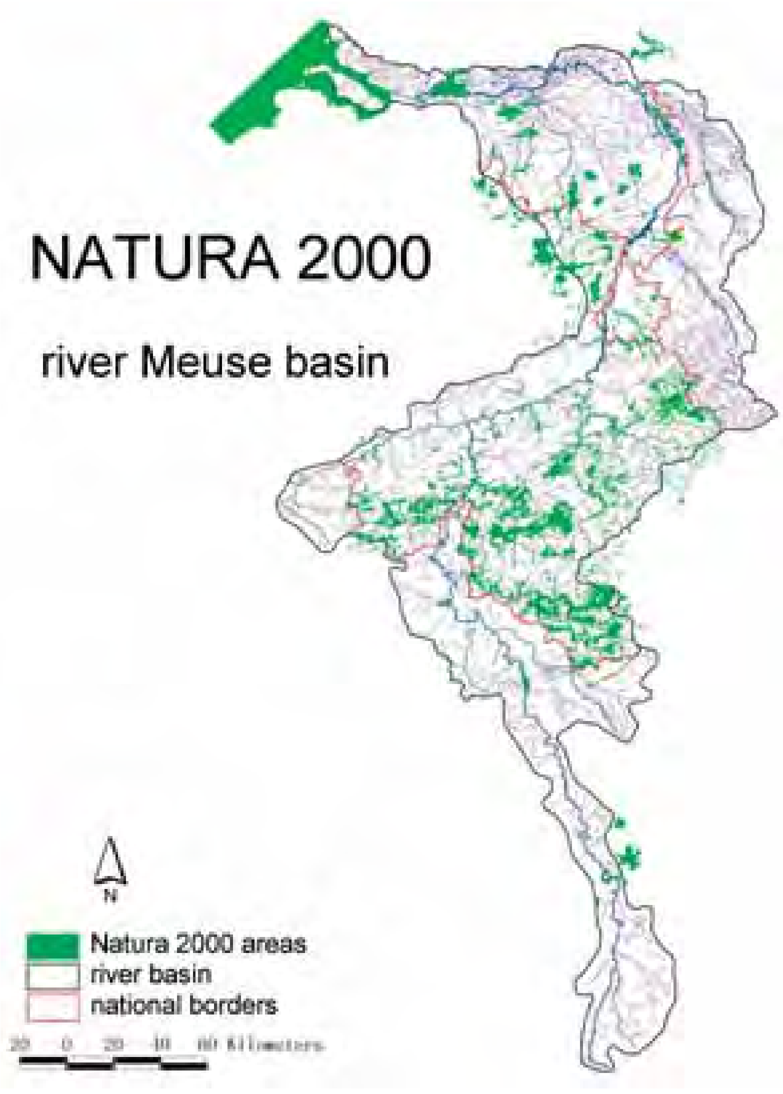


Figure 1.4 Map of water-related Natura2000 protected areas in the Meuse basin.

The Natura2000 map of protected water-related habitats (the German part was unavailable at the date of map construction, 2003) shows the importance of the hydrographical network for the natura2000 network of protected areas. Protection of habitats and species in the Meuse basin is strongly related to the surface waters as many protection zones are situated along tributaries of the Meuse or in the Meuse valley itself.

For our study, emphasis is on the Common Meuse reach, the free-flowing middle course of the river, for 50km bordering Flanders and the Netherlands. It is a gravel-bed river with a strong longitudinal gradient (0.45 m/km) and a wide alluvial plain. The Common Meuse valley consists of a gravel underground with a loamy alluvial cover. The floodplain traditionally was agriculturally used as meadows. Large parts of the alluvial plain have been excavated for gravel mining, leaving large gravel pits or lowered floodplain zones. The degradation of the floodplain natural heritage was the reason to start a river restoration programme and to start local pilot projects, mostly in abandoned gravel mining locations.

For a review of the geomorphology of the Common Meuse valley, we can refer to Paulissen (1972), for hydromorphology to Overmars (1998), Maas (2000) and also chapters IV.1 and V.1. For the hydrology to Berger (1992) and also chapters IV.3 and V.1.

Mapping and sampling

Mapping

In the search for drivers of biodiversity in the river system, an analysis of features and structures is necessary at different scale levels. In our study we distinguished three levels in the river system: the catchment or river basin level, the reach level and the local site level. The choice for mapping units depends on the envisaged applications, and of course the available data and its resolution.

Mapping units for the different scales

Landcover units: the units of the CORINE landcover programme, give a high reso-

lution interpretation of remote sensing data, describing land features, landuse forms and ramifications over grid cells.
Physiotopes: further detail of landcover units, based on landscape generic features and actual abiotic conditions, representing units with homogenous physical characteristics (at the observed scale level; depending on available data from geomorphological and soil maps).
Ecotopes: the smallest distinguishable units, homogenous in ecological (physical and biotical) characteristics, for which biotic characteristics are derived from vegetation data.

Meuse basin mapping

For the river basin mapping, the CORINE landcover units are the best applicable data to make an overall description of land use in the catchment area. Interpretation on land use characteristics in different stretches of the river can be drawn from this map (Jochems & Van Looy 2001). Main distinction in the Meuse river basin is the densely populated northern, downstream part and the less populated southern regions (Ardennes and Lorraine). This affects the intensity of land use and river normalisation (see II.2, III.3 and V.1).

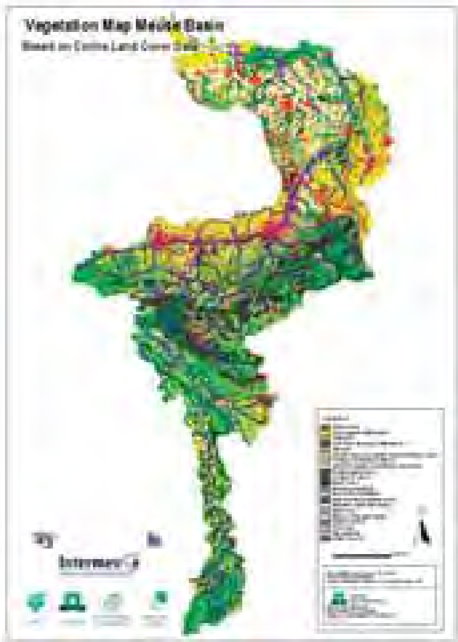


Figure 1.5 Meuse basin map of vegetation classification units based on CORINE landcover data. For a legend description of the Corine, physiotope and ecotope units, see Van Looy & Jochems (2001).

The land use dynamics and the structural aspects of vegetation are important elements on the different scale levels for the interpretation of potential habitat suitability at the regional scale and contact within habitat networks or general landscape connectivity. For the global and regional scale level, the rough distinction of land use units and structural aspects (woodland-grassland, pastures-arable lands) allows an analysis of potential ecological networks on this scale level (Geilen et al. 2002, see V.3).

For the mapping of the reaches, information is required on hydromorphological, geographical, geological and biological characteristics at different scales (see II.2, V.1). A classification of physiotopes, ecotopes and vegetation types was developed for the Meuse floodplains, based on vegetation maps for 3 pilot stretches (Jochems & Van Looy 2001).
The physiotopes, distinguished corresponding to geomorphic and hydrologic units of river valley systems, allow interpretation of the impact of changes in flooding regimes and inundation characteristics. The morphology is the result of local hydrological conditions, induced by the river regime, and therefore the physiotope distinction is based on these characteristics.
For the local level, the soil, inundation and management characteristics are important features for the ecological units legend. Therefore these maps can only be derived from the more detailed cartography of the field, in vegetation or ecotope maps, as was elaborated for the pilot stretches of the Meuse (see Jochems & Van Looy 2001) or for the Common Meuse (see III.1 and III.2).

Common Meuse mapping

For the Common Meuse an ecotope classification was elaborated and a mapping executed for the alluvial plain. The ecotope classification was based on a distinction of local topographic and regional chorological factors (Van Looy & De Blust 1998).
In this way we integrated in the ecotope classification the disorder components of accidental, hazardous dispersal, exceptional hydroregime events and heterogeneous morphodynamics, leading to the major divisions of ephemeral, accidental, fluctuating, contact and low dynamic ecochore series: the first letters in the ecotope type legend of the map.

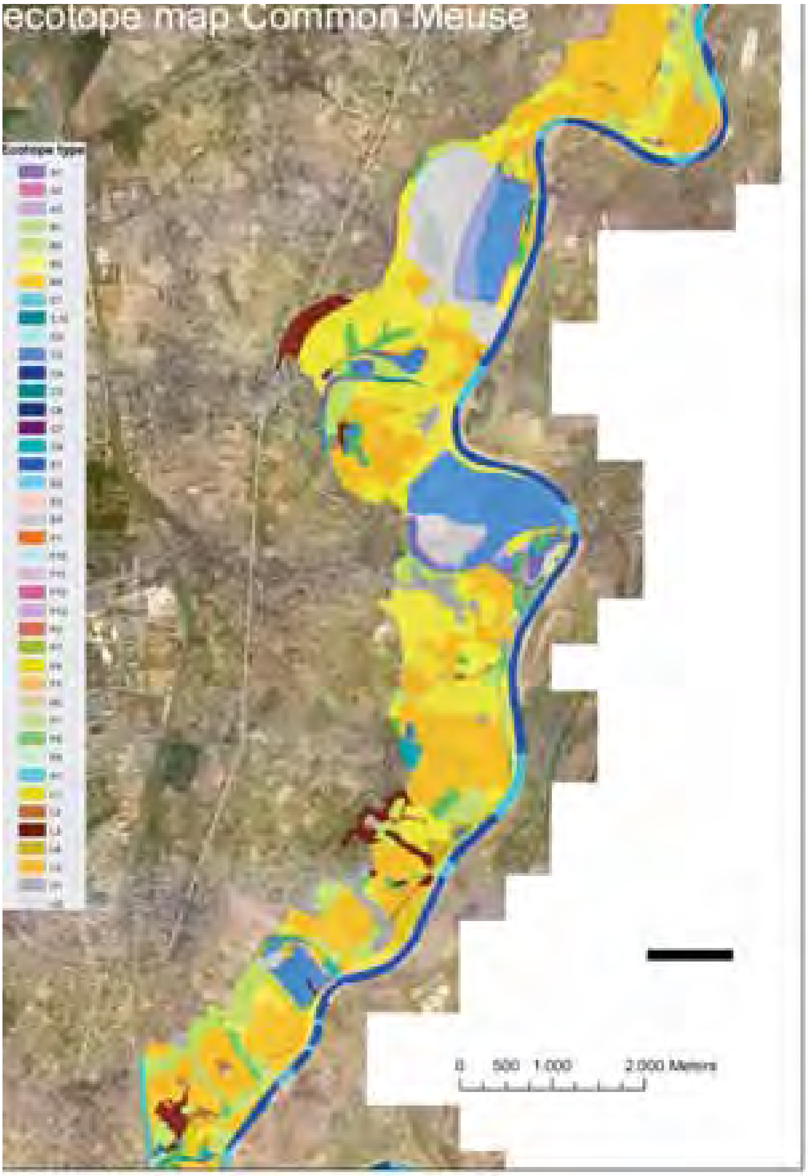


Figure 1.6 Map extract of the Common Meuse ecotope map

Sampling

Sampling was done for vegetation and riparian ground beetles at different scale levels:

1. vegetation sampling was executed within structure categories.

For this choice three elements are at the basis:

- the sampling techniques differ over the structural classes: for pioneer and grassland vegetation 1x1 meter quadrats were sampled, for tall herbaceous vegetation and forest plots 10x10 meters were selected.
- the structural formations are important in management: river management is recently emphasizing on flow resistance aspects of different vegetation structures and repercussions for natural management and general objectives for the floodplains.
- life traits can react differently to environmental factors or changes (see figure 1.3).

Basis for the further exercises was the ecotope mapping combined with vegetation sampling for the Common Meuse alluvial plain in 1999. For each mapping unit a relevee was made at a representative place for the vegetation's composition and diversity within the patch.

Riparian ground beetles

This group of terrestrial riparian invertebrates was investigated for its diversity and spatial and temporal patterns at river basin level, over the Common Meuse reach and with an intensive campaign even up to the level of an individual gravel bar. The international basin sampling of 14 stations was done for one year (2000)(Jochems & Van Looy 2002), the Common Meuse reach sampling was done for two consecutive years (1998-1999) for 19 stations (Vanacker et al. 1999) and the detailed campaign for 2 gravel bars with each 30 pitfalls was executed in the summer period of 2000 for 3 weeks of daily sampling (Lommelen 2000).

Community description

River forest

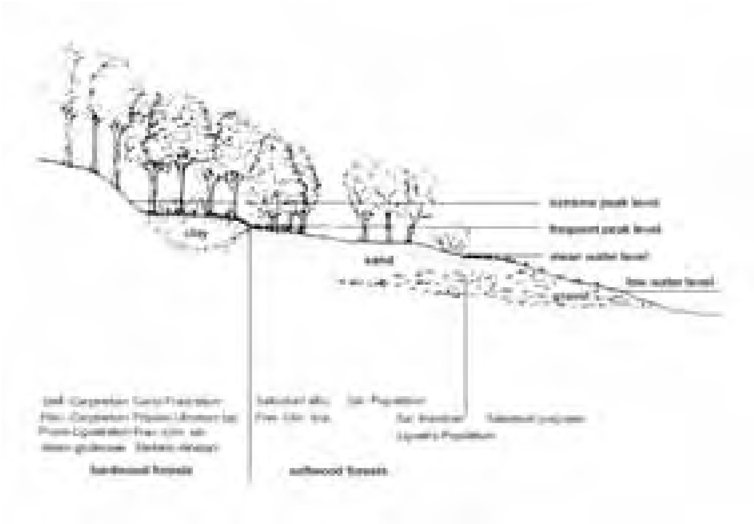


Figure 1.7 River forest types in the gradient over the river-floodplain system.

Typical river forests are the frequently flooded softwood willow, poplar and ash communities. In the present floodplain conditions for the Common Meuse, with it irregular flooding, hardwood forests based on oak can develop. Exceptions are the frequently flooded forest of Hochter Bampd, with different Salicetum and Alno-Fraxinetum communities in development (Van Looy & De Blust 2002), and the riverbank zones and pilot projects for river restoration, where true riparian forests can restore. For the restoration of Black poplar to these developments, a reintroduction is started in 2004 after 3 years of study of the restoration potentials and constraints (genetically) (Vanden Broeck et al. 2002, Vanden Broeck et al. 2004).

Floodplain meadows

Meadow communities for the floodplains show a gradient according to flooding frequency and duration as well. Where in lowland conditions, long-inundated floodplain meadows are a characteristic feature, for the Meuse valley these

are rare, due to the short duration of flood events and the sandy well-drained soils in the area.
The deterioration of the traditional hayfields of the Meuse alluvial plain went on ever more the last 10 years. Yet, the rapid increase and development of species-rich grassland communities in the pilot areas for natural management, puts us in a hopeful mood for safeguarding some remarkable species and communities of the riverine landscape.
Especially the dry calcareous river grasslands are a highly appreciated natural asset of the Meuse valley. The richness of river corridor plants, with their origin in the upstream calcareous regions, gives this community an exceptional aspect for Flanders.

For the other structural categories, typological study and diversity analysis was carried out as well (see fig. 1.3), and published separately: for the tall herbaceous vegetations of the Common Meuse (Van Looy 2002), for the pioneer vegetations (Peters et al. 2000). Furthermore, from this dataset indicator species were derived for the ecotopes and integrated in a monitoring strategy (Van Looy et al. 2002).

Ground beetles

For this community we can show a transect of communities as well, only this is more restricted to a local microhabitat level of the riparian zone, not the entire floodplain. For this group of species, the term assemblages is more frequented than communities. Assemblage classifications exist on the landscape level, for the Walloon region (Dufrêne 1993) and for the Netherlands (Turin 2000). For our habitat assemblage structure, we followed the river habitat templet theory (Townsend 2002) and defined habitat templets for the Meuse riverbanks.

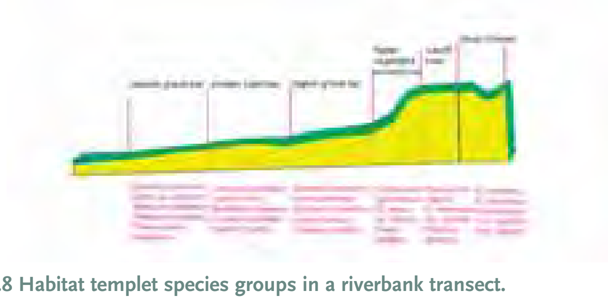


Figure 1.8 Habitat templet species groups in a riverbank transect.

1.5 River restoration approaches and targets

References and targets

In the restoration approaches the quantified target setting is a major challenge. In the context of target setting for biodiversity conservation it is important to refer to the clear definition of restoration options and dimensions. At one extreme, conservation goals reflect the desire to preserve remnants of natural or near-intact systems. Far more common, however, are endeavours to rectify and repair some (or all) of the damage to river ecosystems brought about by human activities. A reference system offers a guiding image for rivers where ecologically sound restoration should be the option, as critical ecological services have diminished (Giller 2005).

The use of references and reference conditions needs a framework in the context of restoration objectives and effective management options. The following definitions of river status guide the selection and use of references: the pristine state of unaltered river systems, with no impact of human activities in the river basin, the natural state of a free river with very limited impact of anthropogenic pressures and morphological processes and contact with the floodplain are intact, although sometimes spatially limited and slightly altered by changed land use in the catchment, and finally the heavily modified or artificial state where human impact on hydromorphological conditions is so strong that the river is far from its natural or pristine state, and the active pressures are irreversible.
Especially the distinction of the pristine and natural state is essential in the scope of the definition of references and targets, according to the Water Framework Directive. As the pristine state of our West-European rivers refers to a situation of lower population pressure in the catchment, with different hydromorphological conditions of sedimentological and general discharge characteristics (Ellenberg 1978), this state offers no perspectives in the definition of reference conditions. The natural state, for many modified rivers refers to conditions before the larger regulation activities of the 19th century and gives a better perspective for the definition of a good ecological status in terms of hydromorphological conditions.
The use of references and the distinction between these conditions is impor-

tant in the scope of the reversibility of anthropogenic alterations. In theory, a pressure is reversible when, in case it is omitted, the system returns to its natural equilibrium state and ecological integrity, corresponding to the reference conditions. Reference conditions should specify the biological potential of a river type within an ecoregion that is minimally influenced by anthropogenic disturbance (Radwell & Kwak 2005). References and reference conditions will be further dealt with in the scope of objective definition (chapter VI.1) and the derivation of hydromorphological reference conditions in relationship to biotic conditions (chapter V.1).

Although the term river restoration has been applied to a wide range of management processes/activities, its precise meaning entails the uptake of measures to return the structure and function of a system to a previous state (an unimpaired or healthy condition), such that previous attributes and/or values are regained (Bradshaw, 1993). In general, reference is made to pre-disturbance functions and related physical, chemical, and biological characteristics (e.g., Décamps 1988; Jackson et al., 1995; Middleton, 1999).

The few studies that have documented geomorphic attributes of relatively intact or notionally

pristine rivers (e.g., Collins and Montgomery, 2001; Brooks and Brierley, 2002, Ward et al. 1999), and countless studies that have provided detailed reconstructions of river evolution over timescales of decades, centuries, or longer (Petss 1989, Piégay Bravard, Décamps et al. 1988), indicate just how profound human-induced changes to river forms and processes have been across most of the planet.

The process of river rehabilitation begins with a judgment that an ecosystem damaged by human activities will not regain its former characteristic properties in the near term, and that continued degradation may occur (Jackson et al., 1995). To this assessment, a clear definition of ecological integrity for a healthy river ecosystem is needed (Karr 1999, Jungwirth et al 2000). The terms in table 1.1 are defined in accordance to nowadays ecosystems descriptions, as they are used to describe the environmental condition and value of rivers (Karr 1999, O'Keeffe, 1997).

Table 1.2 Definitions in target setting.

River health	The condition when a river system's inherent potential is realized, its capacity for self-repair is preserved, and minimal external support for management is needed.
Natural flow regime	Natural conditions of run-off at catchment level, water allocation and flood regime in the river reaches.
Natural baseline	The ecosystems natural (or near-natural) state in biotic conditions and functioning, with a well-defined reference situation.
Ecological integrity	A living system exhibits integrity if, when subjected to disturbance, it sustains an organizing, self-correcting capability to recover toward an end-state that is 'normal' or 'good' for that system.
Hydrologic integrity	Balanced hydrologic, hydraulic conditions on a temporal and spatial scale that are comparable to the natural characteristics of the region.
Biological integrity	The ability to support and maintain a balanced, adaptive community of organisms having a species composition, diversity and functional organization comparable to that of natural habitats of the region.

In our inquiry for reference conditions for the Common Meuse, the Lorraine Meuse was moreover proposed as reference for the Common Meuse (Paalvast 1993), as it offers some interesting prospective/opportunities for defining reference conditions and targets in the context of the WFD. Aquatic communities of the Common Meuse might recover to a level comparable with the less disturbed upstream reach, even through immediate influx of species (Usseglio-Polatera & Beisel 2002). These river reaches are, however, only comparable to a certain degree, for some conditions and/or taxonomic groups (Pedroli et al. 2002). Hydromorphological conditions of the Lorraine Meuse are highly deviating from these in the Common Meuse reach. So, we were obliged to search reference conditions elsewhere or in other terms. As there exists interesting data in historical maps for this region, an ecotope reference condition could be elaborated. For hydromorphological and biological data, actual references were investigated further in the ecoregion (Loire, Allier and Dordogne).

For the regulated Common Meuse reach, the river alterations were already largely present in 1900. The deterioration of biological integrity since is mainly due to intensification of land use. Further hydrological deterioration is caused by embankment, gravel extraction and endikement, resulting in bed incision and distraction of large floodplain area. Biological conservation and restoration strategies often refer to the 1900 reference situation for Western European cultural landscape before industrialisation and land use intensification (Haslam 1996). Proposed measures, classified as mitigation by Boon (1992) concern land use practices and internal management of parcels hydrologic and soil conditions. The natural baseline for these strategies is determined in particular communities and species under specific management regimes of mowing or grazing.

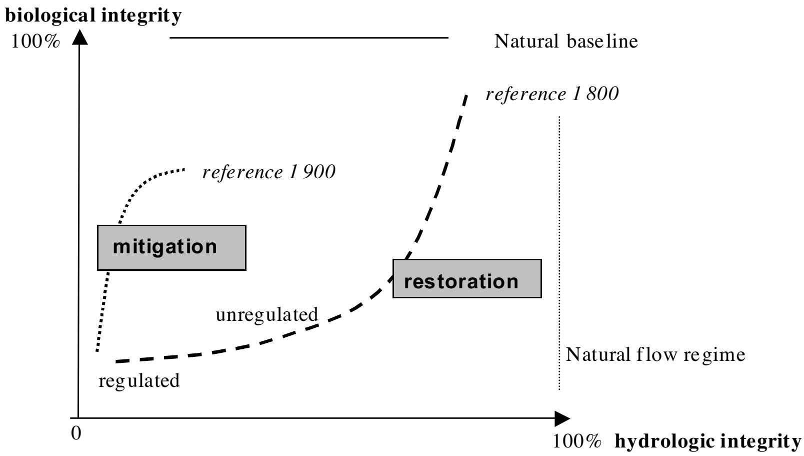


Figure 1.9 Reference conditions and restoration pathways in terms of biological and hydrologic integrity.

In the river restoration strategies, objectives are formulated in terms of processes and target species dependent of river habitat configuration or processes. Measures focus the river processes and mainly the hydrologic integrity. Biological recovery is expected to follow in a spontaneous, non-determined way. This illustration furthermore shows the conservation paradox in choices for river restoration strategies.

The Lorraine Meuse is for most of its course an example of an unregulated river stretch, whereas the Ardennes Meuse, Common Meuse and Sand Meuse are regulated river stretches. Figure 1.9 illustrates the schematic approach of the identification and achievement of a river health condition in terms of hydrologic and biological integrity (goal for strategies/ scenario's) for evaluation assessment at the global and local level.

The natural baseline (maximum habitat/biological/ecological integrity) is achieved when all characteristic communities can have sufficient habitat for the development of sustainable populations. For the Lorraine Meuse, reference conditions are defined in terms of land use conditions of a recent historic situation. The restoration efforts do not imply flow regime conditions, although preservation of this undisturbed character asks many efforts nowadays. Reference conditions in terms of communities and species environment relations need to be interpreted really careful and clearly geographically delineated. Reference conditions for functional groups and hydromorphic processes are applicable in wider ecoregions. For macroinvertebrate communities could be referred to the upstream Meuse reach (Usseglio-Polatera & Beisel 2002), but for hydromorphology and riparian vegetation, reference conditions will be determined over the larger Western plains ecoregion.

Scenarios

The large-scale river restoration project is defined in a master plan for the alluvial plain (see II.1). The concept of the restoration project is to restore hydrodynamics and morphodynamics and related ecological characteristics in a broadened river channel and in re-established secondary channels and backwaters. Planned measures comprise bed widening, bank lowering and side channel reconnection in a comprehensive approach for the river reach.

Three scenarios (figure 1.10) for the river restoration project of the Common Meuse were proposed (Van Looy & De Blust 1995). These scenarios were designed for the Flemish side of the river valley, to join the Dutch proposal for restoration of this border reach (see Van Leussen et al. 2000). For each of these scenarios a description of references and targets was performed, with a persistent and sustainable ecosystem as result. Reference conditions are derived from the historic situation of respectively 1900, 1800 and <1000. Targets are described in distribution and configuration of ecotopes over the valley with specific conditions of management practices for river and floodplain. Although the 'Living river' was decided to be the Flemish spatial planning scenario, these different scenarios and their target definitions still guide us through the choices for specific measures and effect assessments.

Figure 1.10 Flemish scenarios for the Common Meuse restoration project.

1.6 Outline

This thesis contains a number of papers featuring a range of aspects of river restoration and biodiversity conservation topics, brought in the picture at different scales with a set of techniques and approaches for a wide variety of biotic communities, emphasized upon in habitat templets, population dynamic strategies, habitat networks or diagnostic species. Yet, they all tell the same story of a river expressing itself in its unique setting of geomorphology, landscape and biotic features, in a relation that is governed by the flow dynamics.

The trail that a river restoration process needs to follow, with markers and end-points, set out in the multidisciplinary field of engineering and conservationist approaches, is also the outline of our study. We can distinguish the following steps:

- defining scale and area of interest
- conceptualising restoration approach and targets
- determining essential system's functions and processes
- pressure and impact analysis
- deriving effective restoration measures and conservation efforts

- defining the natural baseline/reference condition
- prospecting the restoration potential
- evaluation of efforts
- quantifying targets in a conceptual model
- developing a guiding image to restoration

This brings us to the following flow chart for the thesis:

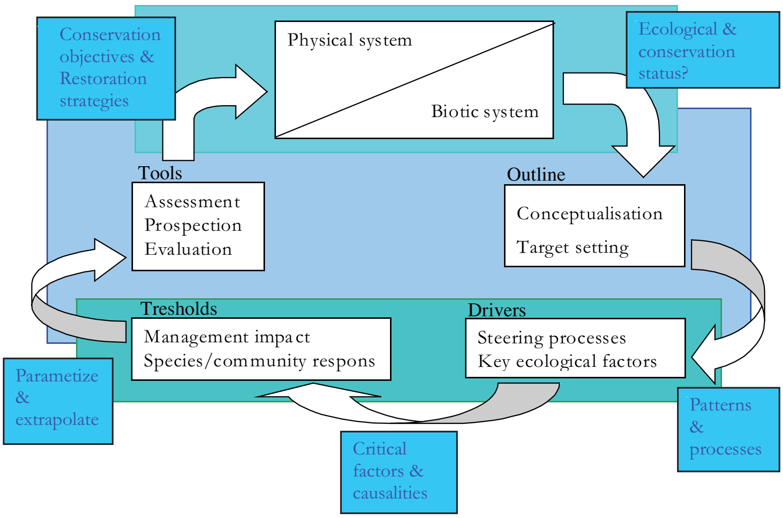


Figure 1.11 Flow chart of information flows in river restoration planning
This flow was structured in the following chapters, as described in Table 1.3, following the structured outline at scale and problem definition from catchment level, zooming to the local sites and microhabitats, back to the river basin approach, with distinct questions and relevant biotic groups and themes.

Table 1.3 Outline of chapter ambitions.

Proces/stage	Scale	question	Theme/group
II. Conceptual framework	Catchment	configuration	Gamma diversity, floodplain meadow, habitat network
III. Steering processes	river, reach, local	heterogeneity	Beta diversity, river forest, floodplain meadow, riparian ground beetles
IV. Pressure-Impact	reach, local	responses, restoration potential	Alpha diversity, riparian forest, floodplain meadow, riparian ground beetles
V. Tools	ecoregion, reach, local	reference conditions, modelling, evaluation,	All groups
VI. Synthesis	All	conservation objectives, Guiding image	All groups

For each of these steps, a clear product (aim) needs to be envisaged, and a relevant organism group or community selected in a specific approach (Table 1.4).

Table 1.4 Analyzed objects and approaches in this thesis.

Object	Approach	Aim	Chapter
River forest	Beta diversity	Steering processes and pressure definition	III.1
	Spatiotemporal sequences modelling	Restoration prediction and quantifying targets	IV.1
	Ecoregion comparison	Reference conditions of hydromorphology	V.1
Floodplain meadows	Gamma diversity	Reaches comparison to define scale and conceptual approach	II.2
	Beta diversity	Dry river grassland steering processes	III.2
	Alpha diversity	Conservation strategies for river corridor species	IV.2
Ground beetles	Alpha, gamma diversity	Habitat conditions and system's processes	III.3
	Beta diversity	Impact analysis and pressure response	IV.3
	Beta diversity evaluation	Quantification and evaluation of targets	V.3
Habitat patches	Habitat network modelling	Targets for spatial configuration	II.1
	Dynamic modelling	Prospection of restoration outcome	V.2

A conceptual framework for target setting in river restoration

In this first chapter, the scope of this study is depicted in its conceptual setting. It gives the outline of the river restoration targets and questions at catchment and landscape level.
The playground and studied objects are presented and their problems analysed from a bird's perspective in a conceptual framework.
This playground is the riparian zone of large floodplain rivers of the temperate region, with the River Meuse as case study.
General aims are the defining of conservation and restoration objectives, and developing effective restoration programmes to reach these objectives. In the perspective of this chapter, the floodplain meadows of the River Meuse and the Common Meuse restoration programme pass the revue.

Surveyed scale levels for this chapter are the river basin and the landscape.

Processes under study are the river's corridor functioning, the connectivity and the resulting identity and coherence in the river basin.

Central questions are: 'What are the drivers for biodiversity patterns in the river system?' and 'What are the triggers for river restoration in a broader view?'

Themes and groups emphasized upon are species and habitat networks, and the floodplain meadows of the River Meuse.

11.1

Setting targets in strategies for river restoration

Pedroli, B., De Blust, G., Van Looy, K. & S. van Rooij 2002. Setting targets in strategies for river restoration. Landscape ecology 17: 5-18.

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Abstract

Since about 90% of the natural floodplain area of rivers in Europe has been reclaimed and now lacks river dynamics, nature rehabilitation along rivers is of crucial importance for the restoration of their natural function. Flood protection, self-purification of surface water, groundwater recharge, species protection and migration are all involved in this process. It is now generally recognised that rivers form natural arteries in Europe but are also of economic importance and are recognisable cultural landscape. Many examples are already available of successful small river restoration projects. Several species thought to be extinct have now reappeared and characteristic species have also expanded in recent years,

This paper concentrates on the concept of setting targets for river restoration as exemplified by the Meuse River. A modelling exercise shows the restraints of current habitat configuration and the potential for habitat restoration along the river. A policy analysis, using a strategic approach, illustrates the influence of the decision making process on the targets for natural river development. River dynamics play a key factor in determining the potential for persistent populations of target animal species along the river, with the help of an expert system (Larch, Landscape ecological Analysis and Rules for the Configuration of Habitat). The potentials for the increase of dispersion and biodiversity and the maximisation of ecological benefits at different scales, are also considered.

Key words: habitat network, Meuse, population persistence, river restoration, setting targets.

1. Introduction: declining biodiversity along rivers

1.1 Decreasing natural dynamics

Rivers have been symbol for the flow of thoughts and prosperity since the origin of man (Schama 1995). They also formed the primary network for exploration and development by man. Rivers are therefore highly modified and adapted to meet the needs of constantly changing societies (Billen et al 1995; Galloway 2000). Major human activities have affected river systems and range from supra-catchment effects to local impacts (Boon 1992). The natural spatial dynamics of many rivers, as well as their temporal dynamics have therefore been altered.

In the lowland rivers of western Europe, engineering works have in general reduced the diversity in habitats and in patterns. The characteristics of flow, that used to be typical for these ecosystems, have now been converted to beneficial conditions for subsistence (Van de Ven 1993). The alluvial landscape is now uniform over large areas, and in many places is only recognisable by the presence of a canalised river, flood levees and a higher density of ditches in the river foreland (Havinga & Smits 2000). These engineering works are designed to control the dynamics of the river, and involve the loss of natural dynamics and of riverfloodplain interactions, as well as the loss of flooding area and fragmentation of habitats.

The dynamics of flow velocity and discharge are key factors in the determination of the fluvial system, and are linked to the suitability of the river as a habitat for biota. Various concepts are used to describe this system, e.g. the river continuum (Vannote et al. 1980), the flood pulse (Junk et al. 1989; Bayley 1991) and hydraulic stream ecology (Statzner & Higler 1986).

From the perspective of the drainage basin and the integrating practice of landscape ecology, the river channel, the river margin and the river floodplain are interdependent and form a single system, referred to as the 'fluvial hydrosystem' (Amoros & Petts 1993; Petts & Amoros 1996). Conditioning processes in these complex fluvial landscapes are related to surface water - groundwater interactions that act in longitudinal, as well as in lateral and vertical directions. An undisturbed hydrology is the precondition for the maintenance of the habitats in their natural state. River bank constructions and flood levees prevent floods that normally lead to the disturbance of hydrology and therefore of habitats, and to changes in ecosystem development. Water management measures have divided the original complex fluvial hydrosystem into a number of distinct, and, almost independent, land units. The original state of interdependent patches has now therefore almost disappeared.

Flooding is the trigger for some of the most important characteristics of a living fluvial hydrosystem as is summarised by the flood pulse concept (Junk et al. 1989). For the river foreland, flooding is the key process determining the pattern and the development of the habitat mosaic. The floodplain therefore presents a lateral zonation regulated by the extent and duration of floods. During flooding, large quantities of water are built up in the alluvial plain. In this phase, energy is dissipated, decreasing the erosive and transporting capacity of the river and keeping the whole river system in a state of dynamic balance. As flood water recedes, so the rivers receive an input of nutrients, contributing substantially to the functioning of the lotic and riparian communities. Urbanisation and the control of water of floodplains for modern agriculture, however, have led to a dramatic decrease of the area available for uncontrolled flooding (Van der Kraats 1994). Furthermore, little of the original storage capacity of the floodplain is left, so that peak discharge control is now most of the time restricted to the river channel itself, compelling to a further impounding of the river (Petts, 1990).

1.2 Fragmentation of the continuous river and riverine landscape system

With running water as the key factor, the river and the adjacent riverine landscape form one continuous fluvial hydrosystem. Engineering works have, however, fragmented this system to a large degree. Weirs, dams and dykes have divided the river into different sections, each functioning almost independently. Habitats in the river foreland are therefore deprived of the essential hydromorphic dynamics (Petts, 1990).

The continuity of the hydrosystem is not only a precondition for its proper, but also makes rivers play an important role in maintaining landscape coherence. From a biogeographical point of view, rivers form a network throughout the drainage basin and provide important pathways for the dispersal and migration of species (e.g. Forman 1995; Reijnen et al. 1995; Foppen & Reijnen 1998). Land use change and river management have destroyed many of the characteristic habitats of the fluvial landscape and hamper their recovery. For many species this means the loss of permanent habitats, temporary functional habitats or stepping stones. Other species are faced with unbridgeable barriers of different types. For example, for many aquatic species a weir is the barriers, whereas for rather mobile riverine species the absence of a patch of softwood in the floodplain within a 10 km stretch may be the problem. Unsustainable populations with numbers of individuals below the 'minimum viable population size' result, linked to impoverished habitats and uncolonisable patches (Chardon et al. 2000).

1.3 River restoration principles to overcome ecological degradation

River restoration seeks to improve the natural functioning of the river and the riverine landscape as a diverse network of habitats, including its corridor function for the catchment. Boon (1992) describes five appropriate strategies for river conservation, in accordance with the state of the river. Where few natural or semi-natural systems with untouched hydrodynamics remain, their preservation is the task. This is rare in Europe, where all large rivers are more or less controlled. For rivers with a still high ecosystem quality and with ecological key factors functioning without major impediments, there the management option is for limitation of catchment development. When the quality is low, their mitigation becomes the case and the development of existing economic and recreational functions need to be accompanied by the implementation of measures that allow the survival of habitats and organisms. When rivers are degraded to a point that natural hydrodynamics are hardly recognisable and only scattered and small remnants of populations persist, there the emphasis shifts towards river restoration. With the help of well chosen restoration techniques and nature development projects, more suitable habitats need to be created, enhancing the recovery of the remaining populations and the establishment of new ones (Gore 1985). The final management option mentioned by Boon (1992), is for the worst case scenario where recovery is hopeless and dereliction is the only wise decision. In these cases, limited resources should not be allocated, but rather directed towards more promising restoration projects.

2. Biodiversity and river management

From the above discussion, it is evident that biodiversity in rivers and riverine landscapes depends largely on the unhampered hydrological and morphological dynamics of the river, functioning in the complex of the 'fluvial hydrosystem' (Amoros and Petts 1993; Petts and Amoros 1996). Fluxes of water, transported components and organisms between distinct environments and spatial units result in a mosaic of interdependent habitats each with characteristic hydraulic conditions, suitable for different species and communities. Any attempt to manage or restore rivers in favour of biodiversity, should focus on these preconditions.

2.1 Longitudinal river diversity

When determining the number of species in river systems and the potential for community recovery, it is necessary to assess the diversity, the quality and the distribution and configuration of the remaining habitats. The habitats in a river system differ gradually from source to mouth, as does the species distribution ('River Continuum Concept', Bayley, 1991). Management and restoration measures should therefore take into account the geographical position of the project site within the river continuum. The selection of a feasible target for the conservation and restoration efforts is then assured as well as their proper adaptation to the prevailing hydrological and morphological dynamics. In most cases however, habitat restoration or development alone, is not enough to obtain environments suitable for sustainable populations. Habitats evolve and their qualities change. The proper qualities can be maintained by applying the appropriate management technique when the habitat is to be controlled by man. However, natural processes can be selected to sustain habitats. In river systems this is achieved when the habitats are still linked to the disturbing hydromorphic processes. Thus, natural succession is hampered or reversed, providing suitable conditions for a huge selection of pioneer species and for species of intermediate succession stages. The diversity and the pattern of habitats and consequently of species, reflect the regime of the current dominant disturbing processes, mainly flow velocity and flooding frequency and duration. These processes operate in a riverine landscape with a characteristic pattern of landforms formed during former phases of erosion and deposition. Features of the river basin such as bedrock type, slope, groundwater hydrology are also important. They determine to a large extent the size and the shape of the river valley and the contribution of groundwater to the total water budget at any site in the floodplain.

2.2 Requirements for persistent populations of animal species

A wide variety of habitats in a range of developing phases is not sufficient to sustain large numbers of species. The population controls of the species and the dynamics of the disturbing processes may indicate the number of comparable habitats needed, as well as their size, localisation and distance from each other. General guidelines for these features can not be given because they depend on the spatial aspects of the population biology of the species concerned (e.g. the area needed to hold a key population, minimal number and arrangement of small habitats to sustain a metapopulation, Verboom et al. 2001), on their dispersal capacity and on the way they use different habitats (Foppen & Reijnen 1998). The objective is to produce a river and an associated landscape, in which barriers and the accompanying isolation no longer put constraints on the free movement and dispersion of species. There must therefore be sufficient suitable habitat, also for colonisation. For species that depend on ephemeral ecosystems and for pioneer species of fast evolving habitats, it is therefore important that the specific habitat forming processes operate in a sufficiently extensive reach of the river and must be in accordance with the appropriate spatial and temporal scales.

3. River identity, basis for target setting

A description of the Upper Meuse in France (Figure 1) shows that traditional and recent values, such as recreation, of the Meuse are integrated to a considerable extent (Pedroli 1999). Such a situation could form a basis for comparison regarding the Common Meuse. The latter currently mainly serves as a discharge channel for water. Recent flood events, however, have proved that the Meuse still is a living river, even threatening damage to newly built houses, enterprises and infrastructure. Currently, new guidelines are therefore being sought for river management and restoration.

Figure 1. The catchment of the river Meuse

The comparison between the two river sections solicits the question as how the river identity can be defined, since the Lorraine Meuse might readily be seen as the ideal reference for the Common Meuse. They are, however, only comparable to a certain degree because the identity of the river is multidimensional. The target images for nature rehabilitation need to consider this multidimensionality, that should be reduced to terms that can be understood by decision makers and politicians.

The described observations together give a firm, yet imprecise, personal impression of the river, which can be ordered by a systematic approach to the identity of the river, starting with appearance, moving into succession and the character as shown below in Figure 2.

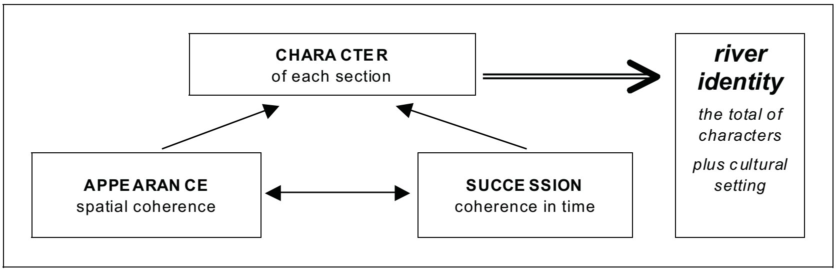

Figure 2. The relationship between appearance, succession and character as stages in the identification of river identity

3.1 Appearance: spatial coherence

Interestingly, a river can not be described from a single point of view. It becomes an image as soon as the observer has combined in his mind the observations of the sites which make it up. The young islands with willow (Salix spp.) seedlings are inseparable from the eroded banks in the next bend whereas the pools and riffles downstream of weirs belong to the same system as the quiet standing water in the backswamps. Some parts of the same section may be sandy, others clayey or gravelly; with steep banks or with gentle slopes. Some flowers may be red and others yellow or green, adding to the image of the same section. These are the phenomena as they appear physically, together forming the spatial coherence. Just like a given tree may produce a richer image when observed from different angles, so the image of the river in spatial coherence is multifaceted.

3.2 Succession: coherence in time

An other dimension is the coherence in time. The presence of plastic bags and straw in the trees along the river indicate that periods of high discharge have taken place. The age of the seedlings on gravel islands indicate past flooding events. The same upper Meuse exhibits many different faces during the day, the seasons, the years. The observed phenomena are continually in transition like the water itself. It requires an active thinking effort to build up a conscious image of this unsteady but none the less characteristic picture.

3.3 Character: the combination of appearance and succession

The character of the river is formed by the combination of appearance aspects and features of succession, that are brought together to give an overall impression. For every section of the river this character is different, resulting in contrasting processes, plants and animals. Upper, middle and lower course can be distinguished, with distinctive plants and animals, water behaviour and banks and floodplains. The composition of these features makes up the recognisable character of a river. The inhabitants of the region can identify the difference between the Lorraine Meuse and the Ardennes Meuse because of the specific composition of their features.

3.4 River identity

It is useful to compare the Meuse with another river such as the Marne in order to identify its principle distinctive features. Comparable physical phenomena and processes are present in both rivers. However, they differ in their overall profile. The Meuse flows through the plains of north-eastern France, before crossing the Ardennes, entering the lowlands and eventually reaching a delta near Rotterdam. In contrast, the Marne has its source in the same area as the Meuse, but then flows through the gentle Champagne hills towards the Paris Basin, where it joins the Seine, which in an estuarine exchange merges with the sea.

The cultural appreciation of a river also determines its individual identity. Whether the river has an influence on society, or vice versa, is subject for discussion (Schama 1995). The characteristics of the Champagne region and its gothic cathedrals undoubtedly give the Marne a different atmosphere than the Meuse which has meadows and fortified medieval churches. At the confluence of the Marne and the Seine, Paris has a major influence on the use of the river, because of its special status for the transport of grain and wine. The lower course of the Meuse is dominated by Liege and Maastricht, and eventually Rotterdam, but river traffic has always been hampered by the gravel shallows downstream of Maastricht. Moreover, the river Meuse flows through the three European states of France, Belgium and the Netherlands. By tradition, each of these countries has specific river management objectives, which have not encouraged integrated development of the river.

Man is inseparably associated with river landscapes. The target images for river restoration need to be realistic in relation to the natural physical processes, and their variation in time, and to the requirements of society has brought about, and which in most instances are irreversible. Even if some of the changes reversed, different situations could result, because of the changed structure of the river. The following section indicates how the above approach could be implemented.

4. The natural river target situation

It is necessary to clearly define objectives when strategies are being determined for the conservation or restoration of rivers for biodiversity. The clear definition of the goals will clarify the types and amount of resources that will be needed for a programme including biodiversity. The area involved in the actions, as well as the problems that have to be tackled and any constraints will determine the chances of success. Such an impact assessment will clarify the position of nature conservation in respect to other societal demands regarding the river. Thus for the proper understanding of the whole river system, an integrated assessment is required before any action is undertaken. Boon (1992) therefore argues that a fifth 'conceptual' dimension should be added to the current four-dimensional description of rivers, comprising longitudinal, lateral, vertical and temporal components (Ward 1989). The definition of the natural river target situation is part of that conceptual dimension (Lenders et al. 1998). It is developed stepwise and is elaborated in the following successive phases.

4.1 The natural river base line situation

Much background information on hydrological dynamics and environmental characteristics of the valley is required to determine what developments can be expected. In the first place it is useful to assemble a vision of the more natural reference that can be adopted as a guideline when designing restoration measures in a particular river (Pedroli & Postma 1997; see Figure 3). In Germany this concept is referred to as the 'Leitbild' (see e.g. Anonymous 1994). It is a description of the desirable stream properties regarding only the theoretical natural potential and not considering any of the economic or political aspects that influence the realisation of the scheme (Kern 1992). As such, it represents the potential for natural development, assuming that human activities in and along the river would cease. In this base line state, the hydrological and morphological dynamics, as well as the associated habitat mosaic, are included. These characteristics will therefore represent the pre-canalisation period from decades or centuries ago, which is mostly the case in European restoration projects. It may also refer to conditions prior to European settlement, as has been stated in the United States of America (Dahm et al. 1995). For the sake of realism, in most of the larger European rivers the presence of flood levees and of controlled discharges must be taken into account when elaborating the expected structure and the processes acting under more natural conditions. They represent irreversible changes in the abiotic environment but also ensure that river restoration, intensive land use outside the floodplain and navigation can go along hand in hand. The existing flood levees then put spatial limits to the restoration projects. The degree of control of the discharge determines the extent to which natural hydrodynamics can act as the driving forces for ecosystem development. Information on the original stream properties can be derived from old maps, photographs and field data and will serve for the definition and mapping of the corresponding habitats or ecotopes, defined as spatially determined habitat types. A hypothetical distribution map of these ecotopes is the result. Pedroli et al. (1996) give a method for this analysis, applied on larger Northwest-European rivers as is shown in Figure 3.

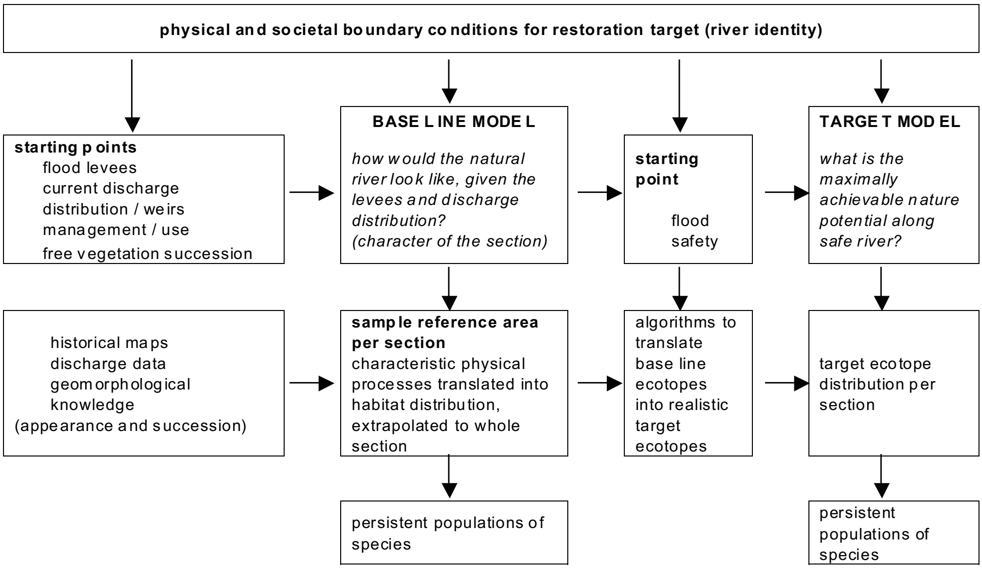

Figure 3. Approach of base line and target models for river nature rehabilitation (after Pedroli et al. 1996)

4.2 Target setting for the ecological state of the river

As stated above, the base line gives a comprehensive but rather hypothetical view. To make it more applicable and suitable for the current planning purposes, it needs to be redefined as a clear target situation for the ecological state of the river; the 'optimal solution' under modified present land use and river use conditions. This target situation results from the combination of the hypothetical base line with the functions of river and riverine landscape that are desired in the future in conjunction with the constraints put on the system by society. In practice safety against flooding of particular parts of the foreland and the maintenance of the transport function of the stream, will frequently be requisites. As a consequence, the control of discharge and of major shifts in the river-bed will continue and the vegetation developing in the floodplain will be managed in order to produce an optimal distribution of successional stages. These will correspond with the storage capacity that is necessary, as well as with the lateral flow characteristics of the floodplain needed to avoid problems elsewhere.

Within the limits set by the hydrological and morphological dynamics of the river stretch, the functions defined and the constraints put by society, there is still a choice of ecological target situations possible. Alternative ecological targets reflect different attitudes towards the role of natural river dynamics or of management activities as the controlling and driving force for nature rehabilitation. In a wider context, there is a problem concerning the human interference that should be allowed in respect to nature rehabilitation. Today, this is a major issue in the debate on the practice of nature conservation and nature rehabilitation. This is especially the case in the intensively used and highly fragmented rural landscapes of Europe, where the ecosystems present are a result of the interaction between man and the environment (see e.g. Arbeitsgemeinschaft Renaturierung Hochrhein 1996). Opinions differ widely and the major restoration projects therefore often start with designing different scenarios introduced in the public debate and presented to the authorities for final decision (Cals et al. 1998). A good example is the elaboration of three strategies for nature rehabilitation along part of the lower River Rhine, each with a specific spatial distribution of ecotopes related to differences in river dynamics and vegetation development control (Reijnen et al. 1995). Another example is the nature rehabilitation along the Common Meuse, as described in the following sections.

5 The River Meuse as an example

5.1 Policy analysis for river restoration

For the preparation of the river restoration project for the Belgian side of the Common Meuse in Flanders, three master plans were elaborated according to different views on the position and the functioning of the natural river in relation to human interference (see Figure 4; Van Looy & De Blust 1995).

- In the first plan, termed Traditional River Foreland, the current distribution and variation of ecotopes is the starting point. Meadows and moderately fertilised pasture, old levees with thermophyllous vegetations, networks of hedgerows, are the significant ecotopes. They are elements in a 150 year old cultural landscape, albeit a, developed after the major impoundment of the river in the middle of the last century. Concern for the species and habitats of this landscape, maintained by low input agricultural management and recently developed recreation activities, restrict the possibilities to re-establish natural hydrological and morphological dynamics. Restoration of the river is for the greater part confined to the ecologically sound restructuring of river banks and gravel pits. Thus, in this view the river ecological functioning depends especially on the traditional use of the river and its foreland.

- In the second plan, termed the Living River strategy, the concept is to restore hydrodynamics and morphodynamics and related ecological characteristics in the primary river channel and in re-established secondary channels and backwaters. Ecotope development will take place mainly along these courses and will yield new habitats for riverine species. The land in between maintains its current functions. During flood periods, the river can expand across the floodplain, penetrating the secondary channels.
- In the third strategy, termed the Free Meuse, the development and the distribution of habitats and species in the whole riverine landscape is considered to be determined by the dynamics of the River Meuse. Within the river foreland there are no restrictions and major human activities are withdrawn. Outside the floodplain and up and downstream of the restored river reach, flooding must be avoided.

At the beginning, the first plan was strongly supported by local nature conservation groups. They considered that it to provide the best chance for maintaining the current biodiversity, whereas the other strategies still had to prove that they could produce high quality habitats. During the further development of the plans, opinions changed. Small scale demonstration projects showed the possibilities for a quick recovery of suitable habitats with characteristic pioneer species after disturbance or creation by the river dynamics. Furthermore, extensive grazing, as a way to maintain the pattern of heterogeneity during succession, turned out to be not only a valuable alternative for the traditional mowing, but also a way to create good germination conditions for the development of new microhabitats (Van Looy & Kurstjens 1997). Today, the river authorities and the government supported by conservation groups have adopted the Living River master plan as the guideline for further nature rehabilitation and river restoration in relation to discharge and flood control of the Common Meuse.

The strategies differ in restoration measure techniques and scale in relation to land use and river dynamics and their reciprocal influence on the development of nature. The most important measures to attain river restoration are channel widening, bank lowering and side arm connection, as shown in Table 1.

Table 1. Main river restoration measures for the three strategies of the Common Meuse floodplain.

Traditional river foreland	Living river	Free Meuse
• Ecologically sound civil engineering in relation to bank protection and dyke construction • Implementation of extensive agricultural management • Hedgerow restoration	• Bank lowering • Secondary channel connection • Implementation of extensive agricultural management • Restoration tributary mouths	• River bed widening • Floodplain lowering • Restoration tributaries

Table 2 gives an estimation of the extent of ecotope groups for the three strategies for the Common Meuse valley. Total area is based on the interpretation of land cover units according the Biological Valuation Map (De Blust et al. 1985) and an evaluation of the strategies (Van Looy & De Blust 1996).

Table 2. Estimation of ecotope distribution (in ha) in the Common Meuse valley in the three strategies.

Ecotope	present	'Traditional River Foreland'	'Living River'	'Free Meuse'
Deep river bed	300	300	250	200
Shallow river bed and gravel bar	50	50	150	350
Secondary channel	0	0	50	0
Softwood forest	39	20	150	250
Hardwood forest	3	3	80	100
Marshland	5	5	105	200
Mosaics of grassland / tall herbs	220	120	800	900
Dynamic shrubland	83	60		
Floodplain ponds	595	650	440	300
Production grassland	710	765	260	15
Agricultural cropping	360	390	40	0

From the transboundary master plan for the Common Meuse that treats the different strategies at length, the Living River strategy was selected by the international Co-ordination Commission as the starting point for future developments (Decision of 1/5/95). It was decided that this strategy, as presented in Figure 4, be assessed for its potential to support a region specific biodiversity.

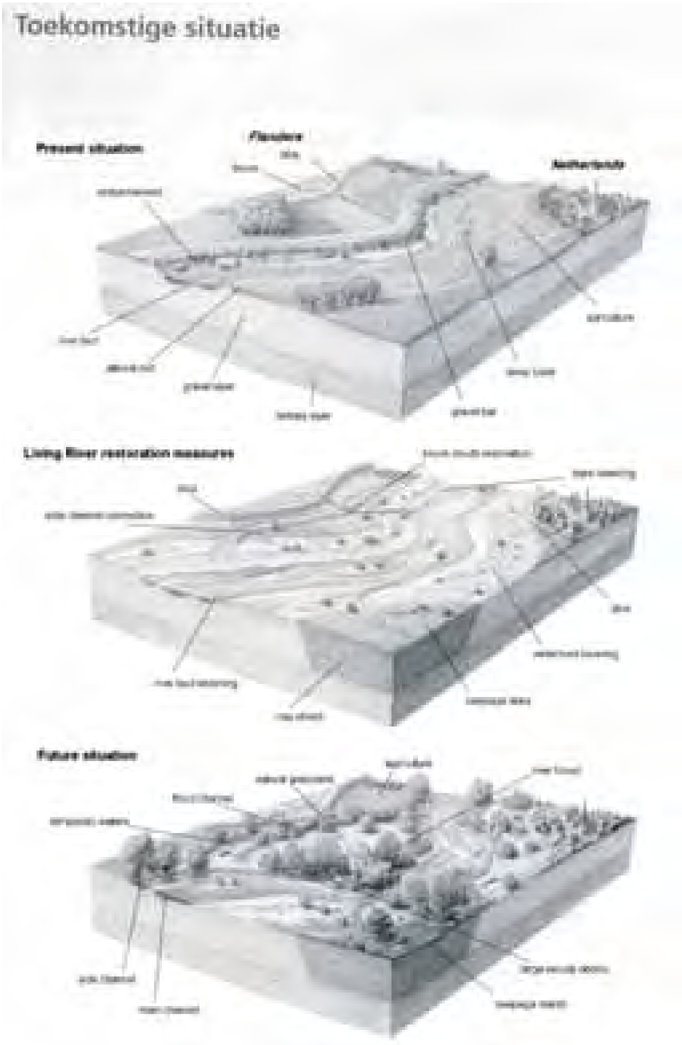

Figure 4. Presentation of the Living River strategy measures and result.

6. Assessment of the potentials for biodiversity

Theoretical and empirical studies have shown that the spatial pattern of a fragmented landscape determines the persistence of natural populations (Vos et al. 2001). In fragmented landscapes, any method for assessment of population persistence or potentials for biodiversity should therefore take metapopulation theory into account (Verboom et al. 2001) and focus on ecological networks. Ecological networks describe the spatial configuration of habitats.
Verboom et al. (2001) propose an approach for assessing ecological networks in which at least one patch (key patch) is relatively large. Based on this assumption and on indices and standards for dispersal capacity of species and population related minimal spatial conditions, an expert system has been developed (LARCH, Landscape ecological Analysis and Rules for the Configuration of Habitat) (Chardon et al. 2000). This system allows the assessment of the persistence of metapopulations in a fragmented landscape and hence can be used to compare strategies that lead to the formation of different landscape and habitat patterns.

For a set of 13 selected species representing certain aspects of natural rives, habitat spatial cohesion was assessed with LARCH for the Living River strategy of the Dutch side of the Common Meuse. The middle spotted woodpecker (Dendrocopos medius), for example, is a model for forest birds with a regional dispersal capacity. For all species, the network of suitable habitats in the flood plain, as well as the habitat network extending into the surrounding areas was assessed on its ability to sustain persistent populations.

Table 3. Some results of the assessment of the spatial arrangement of habitat for selected species in the Living River strategy for part of the Common Meuse (- = negligeable; + = good; ++ = very good).

species	Habitat requirements	Level of dispersal capacity	Potential for key population(s) in plan area	Potential for persistent population in plan area	Potential for persistent population in plan area and surround ing landscape
Barbel (Barbus barbus)	Secondary channels, shallow summer bed	Regional	Yes	-	+
Banded demoiselle (Calopterix splendens)	Shallow open water	Regional	Yes	++	++
Beaver (Castor fiber)	Transition of water and forest	Regional	No	-	-
Gravel spider (Arctosa cinerea)	Gravel and sand bars	Local	Yes	++	++
Blue winged grasshopper (Oedipoda caerulescens)	Gravel bars, grassland mosaics	Local	Yes	++	++
Kingfisher (Alcedo atthis)	Eroded steep banks	National	No	-	++
Common sand piper (Actites hypoleucos)	Transition of water and dynamic shrubland	National	No	-	++
Corn bunting (Miliaria calandra)	Grassland mosaics, production grasslands, crops	Regional	No	-	-
Wood chat (Saxicola rubetra)	Grassland mosaics, production grassland	Regional	No	-	-
Tree frog (Hyla arborea)	Floodplain water and transition to dynamic shrubland	Local	Yes	-	-
Night heron (Nycticorax nycticorax)	Combination of forest, water and marshes	National	Yes	+	++
Natterjack toad (Bufo calamita)	High levees with sandy patches (wintering habitat)	Local	No	-	-
Middle spotted woodpecker (Dendrocopos medius)	Hardwood forest	Regional	No	-	++

The analysis shows that the river restoration strategy indeed offers opportunities for persistent populations of many species, especially for those typical of dynamic river habitats and river foreland, as shown in Table 3. Those species with a regional and national dispersal capacity have their requirements fulfilled regarding habitat cohesion once the restoration measures are put into practice. In addition, the interconnections with habitats up and downstream of the Common Meuse and outside the floodplain, result in a robust habitat network. The model thus stresses the importance of engineering nature development projects in both upstream and downstream sections of the river to produce conditions suitable for persistent populations of many riverine species that function on this large scale. Examples of such species are the middle spotted woodpecker, shown in Figure 5, and the kingfisher. A similar application of LARCH on the Flemish part of the Common Meuse revealed that habitat requirements are met for the beaver (Castor castor), not in the catchment at present, to establish three local populations. The tree frog (Hyla arborea), a species currently under threat, could also develop a core population (Vanacker et al. 1998).

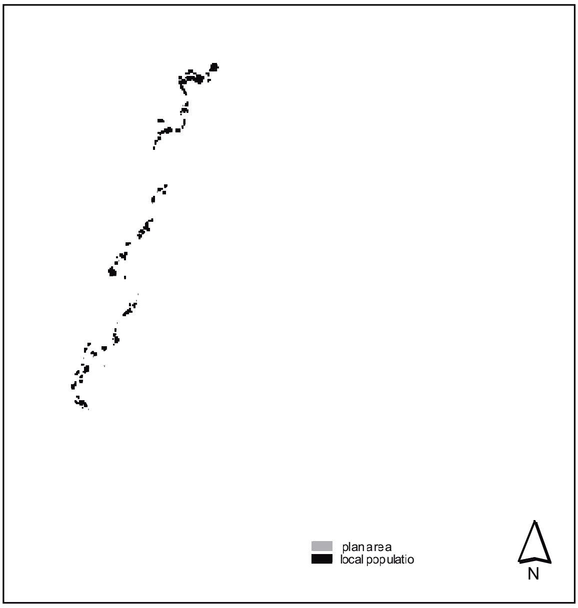

Figure 5. Habitat network of forest species with regional dispersal capacity ('middle spotted woodpecker') in the plan area along the Common Meuse (left) and in the plan area with surrounding landscape (right).

7 Perspectives

Biodiversity: a matter of habitat cohesion

Strategies designed for a large reach of the river and the associated alluvial plain, are rather inaccurate when a precise prediction of the development of a particular site is required. As a consequence, the identification of a specific site of interest, or the exact locations where ecotopes would develop, can seldom be determined. This represents a practical problem because today, in regions with scarce and highly fragmented ecosystems, almost all initiatives and measures to protect and enhance biodiversity are directed towards individual sites. So in order to agree with current policy, it remains necessary in most cases to define the nature conservation values and the ecological functions of the site and to discuss the desired development. Although it makes no sense from a landscape ecological point of view, a particular site, such as an ecotope, is in this respect often appreciated as an isolated entity, which if it is dependent upon the surrounding landscape can cause management problems. The biodiversity present and the possibilities to optimise management activities are necessary criteria for assessment of such sites. Results available from surveys, empirical studies and modelling exercises however, have demonstrated that ecotopes and habitats must be seen as functional parts in ecological networks (Verboom et al. 2001). This is especially true for river corridors (Foppen & Reijnen 1998). The running water itself is an ideal pathway for active and passive dispersion of plant and animal species. The whole riverine system functions as an ecological network, with longitudinal and transversal transfer of water, sediments and nutrients (Petts & Bradley 1997). Migrating animals, especially birds, often use rivers to move through the landscape, where they can also find food and resting places. The considerations presented here, and the Common Meuse example, suggests the potential of further development of habitat network assessment methodology in river restoration studies. Research currently being carried out on the ecology of the whole Meuse will expand these concepts further and will be subject of subsequent publications.

Design with nature....

The Common Meuse example shows that with relatively simple data such as land cover maps and defined criteria of habitat configurations for typical species, can lead to strategies for the development of land use along the river and indicate consequences for natural processes and elements, indicated by animal species. The methodology to define targets for spatial configuration of habitat types, or ecotopes, appeared to be very useful in this context. Instead of concentrating on single habitats, the concept of connectivity is used as a natural guideline to design strategies for nature rehabilitation, because of the interdependence of many landscape elements.

Planning cohesive networks is more effective than conserving species habitats

There are several reasons for changing the species oriented conservation policy into a landscape oriented policy focussing on pro-active strategies:

- Landscapes are the arena for human activities where biodiversity is situated. However, they include many species and habitats combination with different functions, of which nature conservation in only one.

- Many species need different habitats and contrasting spatial conditions. It is therefore not feasible to integrate all species requirements into a single landscape plan. There is a need for integrated planning guidelines for spatial landscape networks.
- The conservation of single species – whether considered as a representative or indicator of other groups – will never be a successful instrument in biodiversity policy when their associated habitats are not considered in their context and configuration in the landscape. Both plant and animal species depend on spatial dispersal – and animal species also on migration – for the long term viability of their populations. Biodiversity planning therefore needs to take account of landscape networks.

These considerations would lead to a policy directed to conservation and development of habitat networks rather than of species or isolated habitats. Specific groups of species should be selected having comparable requirements in the sense of dispersal, migration ranges and barriers. These groups of species may be represented by an idealised key species, for which then sustainable habitat networks can be determined.

II.2

Order and disorder in the river continuum: the contribution of continuity and connectivity to floodplain meadow biodiversity

Van Looy, K., Honnay, O., Pedroli, B. & Muller, S. in Press. Order and disorder in the river continuum: the contribution of continuity and connectivity to floodplain meadow biodiversity. Journal of Biogeography.

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Edited by Jon Sadler

Abstract

Aim

Aspects of connectivity and continuity operating in the River Meuse were analyzed for their contribution to the biodiversity of the floodplain. From this analysis of diversity and composition aspects of the meadow communities, we aimed to derive effective biodiversity conservation strategies.

Location

The River Meuse is one of the larger rivers in the European Western Plains ecoregion. The river's alluvial plains have a long history of cultivation and for these plains the floodplain meadow vegetation is a highly appreciated and valuable nature conservation asset.

Method

We sampled floodplain meadows from 400km of the six geomorphic reaches of the middle to lower course of the River Meuse. For each, 50 vascular plant releves were recorded, representing the spectrum of floodplain meadow communities of that reach. Beta diversity was calculated to quantify similarity in species pools between the reaches. A dissimilarity formula was used to determine the turnover between the reaches and these dissimilarities were compared with a Mantel test to detect whether species composition of the floodplain meadows exhibited connectivity and continuity between the reaches. Species richness for the floodplain vegetation data of the reaches was compared to data for riparian invertebrate communities. The vegetation datasets were ordinated using detrended correspondence analysis (DCA) to reveal patterns in the floodplain meadow species composition and the DCA axes were related to plant functional groups and population strategies. The axis scores of the species and plots were linked to river and plant species traits.

Results

No overall continuity trend in similarity and diversity was observed in a downstream direction. Lateral connectivity was highlighted by the dissimilarity between the reaches and in the influxes of species from adjacent ecoregions. The DCA ordination showed statistically significant separations between reaches and between the plant functional groups. The second DCA axis related to

the river's longitudinal gradient, whereas the first axis showed stronger correlations with river and plant species traits. We termed this axis the 'disorder axis'. Plant invaders and avoiders are located at the left side of the disorder axis, whereas the true river-adapted categories of resisters and endurers are at the right extremity. Contributions to the disorder were identified in terms of connectivity with adjacent ecoregions and the physical disturbance regime of natural and anthropogenic perturbations, resulting in community changes between the reaches.

Main conclusions

We concluded that a single overall strategy for biodiversity conservation of the river or its floodplains is not feasible. Strategies can, however, be derived for separate river reaches based on functional groups in the communities, the reach's disorder characteristics and the influence of surrounding ecoregions.

Introduction

Due to their high habitat heterogeneity and connectivity, river systems are among the most species rich ecosystems in temperate regions (Gregory et al., 1991; Zwick, 1992; Ward et al., 1999). A strong local-regional connection in species pools is thought to exist in river landscapes and explains the high diversity (Naiman et al., 1993; Mouw & Alaback, 2003). Nevertheless, this does not seem to be a general rule, with observed heterogeneity of species diversity in riparian landscapes leading several authors to conclude that studies should focus on understanding the causes of variation in species richness (Nilsson et al., 1989; Sabo et al., 2005).

Understanding patterns of distribution and abundance of species requires that we test theoretical predictions about functional relationships between species and their environments across a range of spatial and temporal scales (Poizat & Pont, 1996; Poff , 1997). River ecology concepts deal with these observed patterns in biodiversity. Research that deals with longitudinal changes in biodiversity uses the River Continuum Concept (RCC) as a theoretical framework for hypothesis testing (Vannote et al., 1980, and see e.g. Statzner & Higler, 1985; Oberdorff et al., 1993; Grubaugh et al., 1996). The RCC indicates that gradual changes in communities along the river are a product of resilience strategies, expressed in functional adaptations (Naiman & Décamps, 1997; Shafroth et al., 2002) or metapopulation dynamics (Bunn & Hughes, 1997), that result in an increase of shared species pools and species strategies (Tabacchi et al., 1996; Sabo, 2005). As the river corridor is seen as a continuum, similarity increases gradually downstream. Evidence of the river corridor functioning for metapopulations, gene flow and species dispersal (Gouyon et al., 1987; Johansson et al., 1996; Henry et al., 1996; Andersson et al., 2000; Imbert & Lefèvre, 2003), indicates the overall importance of connectivity and continuity in the river corridor for regional biodiversity.

Other riparian floodplain concepts deal with the lateral relations, perpendicular to the river course. Dynamics concepts like the Flood Pulse, Flow Pulse and Patch Dynamics concept (Connell & Keough, 1985; Junk et al., 1989; Large & Petts, 1996; Tockner et al., 2000) indicate functional relations between the river and the vegetation mosaic in the floodplain, with diversity patterns resulting from community and species responses to disturbances. The functioning of river corridors has also been considered in terms of their discontinuous variation in space and time; the concepts of Shifting Mosaics and the Intermediate Disturbance Hypothesis (Tabacchi et al., 1990; Malanson, 1993; Cooper et al., 1998). These concepts predict the effect of disturbance on diversity patterns, as species diversity has been found to be highest at intermediate levels, along disturbance gradients (Huston, 1979).

Linking diversity in species numbers and life-forms to environmental gradients and general landscape features can also be an important guide in the development of conservation and restoration strategies (Wang et al., 2002). Recent publications suggest the use of classifications of plant strategies for understanding the processes that structure species composition and diversity in the river corridor. Examples are classifications into specialists and generalists (Mouw & Alaback, 2003), functional adaptations to river dynamics (Naiman & Décamps, 1997; Townsend et al., 1997; Turner et al., 2004) and analyses of regional scale population dynamics (Freckleton & Watkinson, 2002). Floodplain meadow vegetation is an interesting type in this respect, due to its richness in species and variation in plant strategies (Girel & Manneville, 1998; Leyer, 2005). A scientific research programme, in a collaboration of institutes of the three bordering countries along the Meuse river, developed an evaluation approach for the river's biological integrity (Geilen et al., 2004) and a key element of this approach was an understanding of the dynamics of floodplain meadow vegetation. Here we test the following contrasting hypotheses, based on river concepts, which focus on the differences between longitudinal and lateral relations:

1. Continuity in the river system results in gradual changes in communities and increasing similarity in a downstream direction.
2. Lateral connections lead to discontinuities in downstream changes.

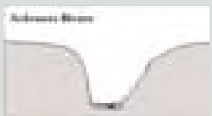
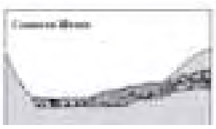
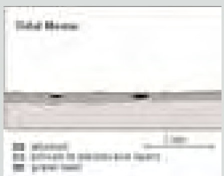
Materials and methods

Sites and sampling

The River Meuse is a rain-fed river, originating at an altitude of 409 m above sea level on the Plateau of Langres in the Northeast of France and discharging into the North Sea some 900 km further downstream. The catchment area of the river is 34,500 km^2 and is situated in France (9,500 km^2), Belgium (14,500 km^2), Germany (4,000 km^2), Luxembourg (600 km^2) and the Netherlands (6,000 km^2) (van Leussen et al., 2000). The Meuse was divided into six reaches based on geomorphological and hydrological characteristics (Pedroli & De Leeuw, 1997) (Figure 2.6, Table 2.4). The Lorraine reach has a very narrow limestone basin with a very wide river valley. Crossing the Ardennes rocky formations, the river is forced into a narrow valley with high rocky slopes. The valley widens again in the Common Meuse reach, where the river has deposited a thick gravel layer. The river becomes less dynamic in the lower sandy reaches. The Peelhorst reach has a rather narrow valley in the sandy Pleistocene terraces landscape and conditions become more neutral with the inflow of acidic waters from the Peel moors region. The valley then widens a little as the river turns west and enters the Holocene Rhine-Meuse interfluvium. Further down, the estuarine tidal impact becomes tangible in the Tidal Meuse with a further broadening of the river system.

Figure 2.6 Map of the Meuse basin showing the sample reaches and ecoregions.

Table 2.4 Floodplain meadow soil and hydromorphological characteristics for the Meuse reaches. Stream power $\Omega=\gamma.Q^{bf}.S$ (W/m), where $\gamma=\rho.g=$ 6.25, Qbf= bankfull discharge and S= slope (Brookes, 1988). Flow variability in Coefficient of Variance= $\partial Q/\partial t$ (Growns & Growns, 2001) for daily discharge data from gauging stations in the middle section of the river (Fig. 1). Groundwater amplitude is in metres.

Reach and main tributaries	Geomorphological profile	Geo-/hydromorphological variables	Floodplain soil stand conditions
Lorraine Meuse Km 200-280 *Mouzon*		Stream power: 500 Valley width: 200-2000 Discharge ampl.: 5-500 Flow variability: 35	Groundwater ampl.: 0.5 Soil pH: 6.5 ± 1.5 pH amplitude: 3
Ardennes Meuse Km 280-450 *Chiers* *Semois* *Sambre* *Ourthe*		Stream power: 750 Valley width: 100-300 Discharge ampl.: 10-800 Flow variability: 10	Groundwater ampl.: 1.1 Soil pH: 6.5 ± 1 pH amplitude: 2
Common Meuse Km 450-500 *Jeker* *Geul*		Stream power: 5000 Valley width: 300-2500 Discharge ampl.: 10-3000 Flow variability: 70	Groundwater ampl: 3-5 Soil pH: 6.5 ± 2.5 pH amplitude: 5
Peelhorst Meuse Km 500-580 *Roer* *Swalm*		Stream power: 1750 Valley width: 300-800 Discharge ampl.: 20-3000 Flow variability: 15	Groundwater ampl.: 1.2 Soil pH: 5 ± 1.5 pH amplitude: 3
Sand Meuse Km 580-670 *Niers*		Stream power: 1625 Valley width: 800-2000 Discharge ampl.: 50-3000 Flow variability: 15	Groundwater ampl.: 0.9 Soil pH: 5 ± 1 pH amplitude: 2
Tidal Meuse Km 670-750		Stream power: 1125 Valley width: 1000-5000 Discharge ampl: 100-3000 Flow variability: 20	Groundwater ampl.: 0.7 Soil pH: 6 ± 1 pH amplitude: 2

System descriptions and historic survey records (Van Dijk et al., 1984; Duvigneaud & Saintenoy-Simon, 1992; Grévilliot et al., 1999) were consulted to locate important floodplain meadows in defined study reaches. Fifty floodplain meadow vegetation releves were selected within each of the six defined reaches. The releve sites were situated in the centre of the reaches, near river gauging stations (Figure 2.6). Many of them are in protected areas or nature reserves and were dispersed across a variety of environmental conditions using hydrological and management conditions to stratify the samples (Grévilliot & Muller, 2002). Vascular plants were sampled and determined up to species level in each releve using the Braun-Blanquet method (1x1m quadrats).

A range of environmental variables were gathered from the releve sites and the nearest gauging stations (Table 2.4). The hydrological variables of the river flow's Coefficient of Variance ($\partial Q/\partial t$) and discharge and water level amplitude as measures of hydromorphological dynamics were derived from 10-year flow data from the gauging stations according to standard definitions in the literature (Growns & Growns, 2001). The abiotic data of the plot sites were collected during the field surveys; the pH from topsoil sampling using Metrohm (titration, pH-carrousel), and the groundwater amplitude from piezometric data. A global matrix was constructed for the vascular plant data, with abundances over the plots in the reaches determined (s:1-5%, r: 5-10%, o: 10-25%, f: > 25%) (see Appendix S1 in Supplementary material).

Two sets of faunistic data were consulted from River Meuse surveys with the same aims as our study; to compare richness and composition of the riparian communities along the river. The sets are described more extensively in publications; for the terrestrial carabid beetles (Van Looy et al. 2005) and for the aquatic macroinvertebrates (Usseglio-Polatera & Beisel 2002). The riparian ground beetle sampling was executed in 2000 using 14 stations along the River Meuse, each consisting of two rows of three pitfalls in the riverbank. The traps were sampled at two-week intervals for the period May to October. Samples from the three traps were pooled and species identified in the laboratory. The macroinvertebrate data were gathered form a standardised sampling protocol in an international Meuse monitoring programme for 16 macroinvertebrate sampling locations. In August 1998, faunal samples were simultaneously collected from banks and the channel (either dredge samples, surber or hand net samples, or artificial substrates of pieces of brick in polyethylene netting for 28 days in-situ). Identification of extracted ground beetles was determined up to species level, and for macroinvertebrates to species or genus level (except for Diptera, for which sub-family and family levels were used, and for Nematoda, Oligochaeta and Hydracarina, which were recorded as such). Taxonomic richness of the sampling stations was compared over the reaches.

Diversity and similarity analyses

The Shannon index was calculated to determine the diversity in species richness and composition of the floodplain meadow vegetation, which can be considered as a measure for the level of disorder in biodiversity analysis over geographical regions (Orloci et al., 2002).
Beta diversity was calculated to quantify similarity in species pools between the reaches (Schluter & Ricklefs, 1993). We used a dissimilarity formula as the metric of beta diversity, to determine the proportion of species in common ($\beta = \sum$ unique species in each reach / $\sum$ of all species in both reaches). We used a Mantel test to measure and test the linear correlation between two matrices, a dissimilarity matrix derived from the plant species, and a distance matrix using geographical distances between the sites. Observed patterns in species richness for the reaches were compared with results of other studies of longitudinal changes in riparian communities along the Meuse (Usseglio-Polatera & Beisel, 2002; Van Looy et al., 2005).
The vegetation datasets were ordinated with detrended correspondence analysis (DCA) using CANOCO 4.0 (Gauch, 1982; ter Braak and Smilauer, 1997).

Functional grouping

Naiman & Décamps (1997) proposed a classification of riparian plants into four broad categories of functional adaptations: (1) Invaders, which produce large numbers of wind or water-disseminated propagules that colonize alluvial substrates; (2) Endurers, which are well adapted to living under a number of disturbance regimes; (3) Resisters, which are less broadly specialized river-adapted species, resisting specific stress conditions with a specific strategy;

and (4) Avoiders, which lack adaptations to river dynamics. We applied this classification based on the species competitive and regeneration strategies (Table 2.5), with an additional screening for species traits such as number of seeds and dispersal capacity, as documented in the Flemish floristic register (Van Landuyt, 2001), a database of species distribution data and species traits.

We distinguished ecoregions in the Meuse basin (Figure 2.6) – delineated geomorphologically – using the phytogeographic regions definition (Delanghe et al., 1983). The Primary mountainous formations of the Vosges, Ardennes and Eifel districts contain the catchment's strong siliceous peaks, and these are crossed by several smaller calcareous regions with softer rocky soils, allowing the widening of the alluvial plain. The lowland reaches flow across Tertiary sandy and loamy regions and Quaternary alluvial plains and delta deposits. For the analysis, species were appointed to specific ecoregions based on their classification into socio-ecological groups (Stieperaere & Fransen, 1982), categorized according to abiotic conditions (soil texture, humidity, acidity, salinity) and adjusted with the distribution data of the floristic registers (Delanghe et al., 1983; Van Landuyt, 2001; Van Rompaey & Delvosalle, 1979).

Table 2.5 Classification of the functional groups of plant species based on plant species traits documented in the Flemish floristic register (Van Landuyt, 2001).

	C-S-R strategy*	regeneration strategy†	dispersal agent‡
avoider	-	S, Bs	unsp/anim
invader	C-S	S+	Wind
resister	S or R (max 1)	V/S	water/unsp
endurer	C-S-R (min 2)	V	water/wind

* C-S-R: competitor, stress-tolerant or resister, † regeneration strategies: S: seed, Bs: seed bank, S+: many seeds, V: vegetative, ‡ dispersal agents: unsp: unspecified, anim: animal dispersed.

A general species amplitude classification is present in the floristic register (Van Landuyt, 2001) based on the territorial typology of the physical environment. The generalists are species with higher amplitudes, i.e. less sensitive in terms of soil and hydric conditions.

To classify the species population dynamics in the Meuse river system, the Freckleton & Watkinson typology was translated into a scheme of species and patch criteria (Table 2.6). The strategies were attributed based on species frequency and abundance in the plot-species matrix and the floodplain vegetation mapping in the international project (Geilen et al., 2004). The main distinction is between regional and local populations. In terms of the application of metapopulation theory, regional populations are those relying on colonization from upstream populations. The species were assigned to one of these strategies without the evidence of a lengthy population study and no reference was made to current discussion on the distinction of metapopulations in non-continuous habitats and the evidence for extinctions and discrete habitat patch use (Gouyon et al., 1987; Ouborg, 1993; Eriksson, 1996; Freckleton & Watkinson, 2003). Nevertheless, this generalized strategy interpretation offers interesting opportunities to analyze aspects of species dispersal at a regional scale (Freckleton & Watkinson, 2002).

Table 2.6 Classification of population dynamics types. Data were compiled from species frequency and abundance in the plot-species matrix and the floodplain vegetation mapping in the international project (Geilen et al., 2004).

Population dynamics type	Source population, immigration	# populations	Abundance within patch	Patch type selectivity	Patch frequency	Occupation of suitable habitats	Patch dynamics	Patch size, isolation
Metapopulation	upstream	rare-occasional	rare-	High occasional	frequent	partially	low	small, dispersed
Source-sink	upstream	rare-occasional	frequent	Low	frequent	low	high	-
Remnant population	local	rare-occasional	rare-occasional	High	rare-occasional	partially	low-high	small, isolated
Shifting cloud	local	rare-frequent	rare-occasional	Low	frequent	low	high	-
Patchy population	local	rare-frequent	frequent	High	occasional	high	medium-high	small, dispersed
Extended local population	local	frequent	frequent	Low-high	frequent	high	low	large

The DCA scores derived from the vegetation data were related to the species amplitude by Mann-Whitney tests, and to the classifications in ecoregions, functional adaptation categories and population dynamic strategies with a Kruskal-Wallis test. This was followed by one-way Tukey pairwise comparisons between the categories.

Results

Diversity and similarity

From the vegetation survey, 407 species of vascular plants were recorded in more than one releve and were attributed to the different classifications (see Appendix S1). The Shannon index of 1.71 for the floodplain meadow plots indicates a high diversity in composition and species richness of the floodplain meadow vegetation or a high level of disorder present in the dataset of the floodplain meadows compared to other studies (e.g. Ward, 1998; Deiller et al., 2001; Orloci et al., 2002; Wang et al., 2002).

The Mantel test (Table 2.7) shows a significant correlation (0.63) between the dissimilarities and distance between the reaches (p=0.004). Similarity between adjacent reaches was substantial, but no gradual shift was identified in similarity over the reaches (Table 2.7). Remarkably, for reach I the highest similarity is with reach VI. Table 1 also indicates the similarity between reaches I and VI, for groundwater amplitude and flow variability. The presence of floodplain meadows that are inundated frequently and for prolonged periods distinguish these reaches from the others.

Table 2.7 Dissimilarity values (∑ unique (i.e. unshared) species in each reach / ∑ of all species in both reaches) for the floodplain meadow plant data between the reaches and distances between the reaches, as introduced in the Mantel test, with the correlation result at the foot of the table. Based on 50 1 x 1 m releves per reach.

	I	II	III	IV	V	VI
I						
II	0.497					
III	0.56	0.24				
IV	0.52	0.42	0.32			
V	0.52	0.5	0.45	0.31		
VI	0.45	0.54	0.52	0.39	0.34	

	I	II	III	IV	V	VI
I						
II	1					
III	2	1				
IV	3	2	1			
V	4	3	2	1		
VI	5	4	3	2	1	

0,626222 correlation

The species richness peaks in the Common Meuse reach (III) for the floodplain meadows (Figure 2.7). For the aquatic and terrestrial riparian invertebrates the furthest upstream reach shows the highest species richness. The peak in the Common Meuse reach is also present in the diversity of plot species richness for the floodplain vegetation and the riparian ground beetle communities (Figure 2.8).

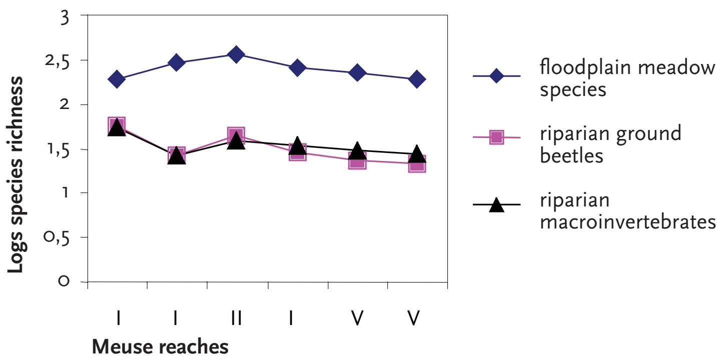


Figure 2.7 Logarithmic value of species richness over the reaches; from total taxonomic richness over 315 floodplain meadow releves (total species richness: 407), 14 ground beetle sampling stations (total species richness: 86) and 16 macroinvertebrate sampling stations (total species richness: 117). Log values are used for this transformation to allow clearer interpretation, as overall group species richness differs strongly.

Functional groups

The functional adaptation classification of floodplain species was significant related to the first DCA axis (Chi-square 9.57, $p=0.023$). Along the axis the invaders are situated to the left, whereas the true river-adapted categories of resisters and endurers are at the right extremity (Figure 2.9). The avoiders take an intermediate position on the first axis, more differentiated by the second axis, as they are concentrated in the upstream reaches. For the second axis, only marginally significant differentiation between functional groups was found (Chi-square 7.27, $p=0.064$). The adaptations of avoiders and resisters are of most benefit to survival in the floodplains of upstream reaches, whereas endurers are more common downstream.

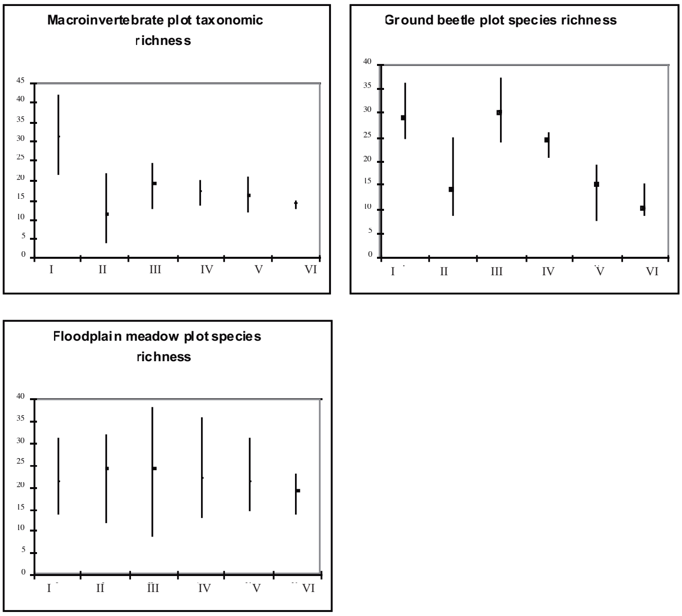


Figure 2.8 Changes in species richness for the samples of the three biotic groups over the Meuse reaches (mean, minimum and maximum values). Data from international Meuse monitoring programmes (Liefveld et al. 2001, Usseglio-Polatera & Beisel 2002, Van Looy et al. 2005).

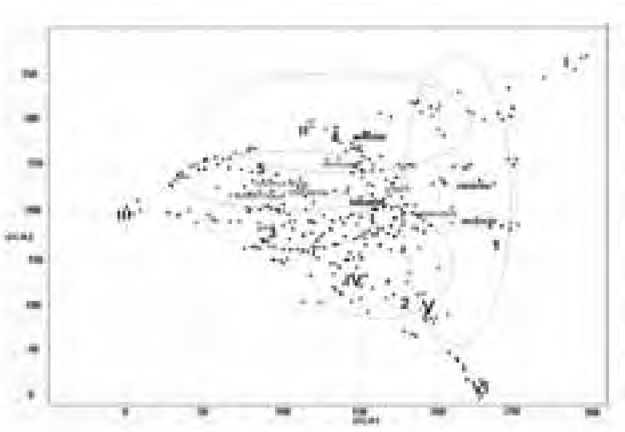


Figure 2.9 DCA-plot of plant species and mean scores for the different classifications of functional and population dynamic traits (indicated in words). The roman capitals indicate the River Meuse reaches; the numbers and confidence ellipses indicate the ecoregions (1: Fluvial region, 2: Campine sandy region, 3: Brabant loamy region, 4: Calcareous region, 5: Primary regions).

The ecoregions classification was significantly related to both the first and second axis (Chi-square values of 72.87 (p<0.001) and 57.73 (p<0.001) respectively). Figure 2.9 illustrates this connectivity of the reaches and adjacent ecoregions within the river catchment, a trend that is also clear in the ecoregions partition diagram (Figure 2.10), with the species input along the river of ecoregions 2, 3 and 4. Reaches I and VI also show a high level of similarity in this diagram. The most species of adjacent regions are present in the midreach.

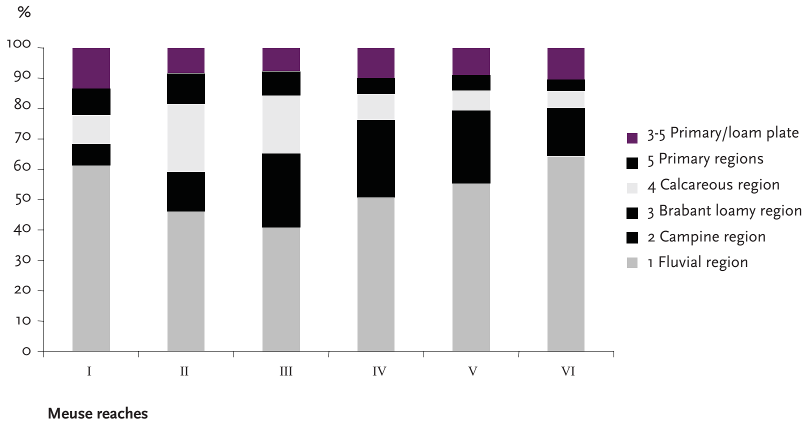


Figure 2.10 Species ecoregion classification for the plant species of the floodplain meadow vegetation for the Meuse reaches (see Appendix S1).

The amplitude classification tested significantly for the first axis (Z: 2.2, p:0.025), but not for the second axis, as did the population dynamic strategies (Kruskal Wallis Chi-square test result: 17.07, p<0.001), suggesting a grouping of source-sink, shifting cloud and metapopulation strategies to the left side. In contrast, remnant, patchy and extended local population strategies are located more to the right. The one-way Tukey pairwise comparison revealed significant differences (p<0.05) between extended local populations and patchy populations on the one hand, and the source-sink, shifting cloud and metapopulations on the other. The remnant populations are only significantly separated from the shifting cloud strategists and are situated at the extremity of the first axis. However, on the second axis they were clearly separated from the extended local populations, with the remnant populations concentrated more upstream, in contrast to the extended local populations dominating downstream.

Discussion

Diversity patterns and disorder

The diversity in our dataset is high compared to other studies (e.g. Ward, 1998; Deiller et al., 2001; Orloci et al., 2002; Wang et al., 2002). For this high disorder in species richness and composition of riverine communities, we identified a variety of sources.
Since the contribution to disorder from hydroregime, management and surrounding ecoregions differs over the reaches, the dissimilarities between reaches can be high. At the River Meuse basin level, the dissimilarities between the floodplain meadow vegetation of the reaches are quite high and no longitudinal trend in similarity is present, thus refuting the first hypothesis. Our data suggests that continuity of community changes along the river, as identified in the River Continuum Concept (Vannote et al., 1980), is not a continuous pattern nor does it display a simple relation to river scale (Statzner & Higler, 1985; Ward & Stanford, 1995; Grubaugh et al., 1996). Similarly, Tabacchi et al. (1990) described discontinuity in the longitudinal changes of riparian vegetation composition and reported high species turnover between reaches of the River Adour. Rapid changes were explained in their study by transitions from higher to lower altitude and tributary influences. For the Meuse, changes were primarily related to reach characteristics of flow variability, valley form and soil conditions, resulting in significant differences in the population strategies present, and were also observed in the presence of species from the surrounding ecoregions.

The lateral connectivity predicted in the second hypothesis was observed in the dissimilarities between the reaches, as well as in the ecoregion classification. Connectivity works throughout the river basin, among the reaches, but it was also significantly highlighted in the lateral relationships with surrounding ecoregions. Dynamics concepts in this context focus mainly on the connectivity as an expression of the disturbance adaptations of communities, rather than regional scale (population) dynamics relating to contact with the hinterland. This contact is not necessarily an expression of disturbance but can just as easily occur in extended local or patchy populations, especially in wider valleys with good contact to surrounding uplands. Several studies have identified spatial population dynamics that did not correspond to changing physical environmental conditions but resulted from local dispersal and survival strategies (Freckleton & Watkinson, 2002;

Dirnböck & Dullinger, 2004; Jäkäläniemi et al., 2005). Our ecoregion and regional population dynamics classification approach was very successful in highlighting these aspects and revealed that lateral relations in the river basin are just as important in explaining the diversity patterns as longitudinal ones.

The pattern of highest diversity in midreaches, described in the River Continuum Concept, is also observed in our data, but as Statzner & Higler (1985) pointed out, no general correlation to order and no ecological argument can be found to generalize this relation. The same conclusion counts for the Intermediate Disturbance Hypothesis, as there are aspects at the reach level confirming this theory (Van Looy et al., 2003). Yet, at a river basin level, biodiversity is maximal in the reaches with maximum disturbance, and not intermediate. The scale-sensitivity of the Intermediate Disturbance Hypothesis has been stressed by Pollock et al. (1998), who noted that small-scale patterns of spatial heterogeneity influence biodiversity patterns, yet no link to large-scale intermediate disturbances can be made.

The observed diversity pattern for the floodplain meadow vegetation shows resemblance to the diversity of terrestrial and riparian invertebrate communities of the River Meuse. As the riparian fauna diversity is closely related to habitat conditions, influenced by bank management (Usseglio-Polatera & Beisel, 2002; Van Looy et al., 2005), the floodplain's biodiversity gives a broader picture of influences present in the river system. The upstream reach is mostly unregulated and has the best riparian habitat quality. Therefore, the high diversity in the Common Meuse reach is the more striking, as this reach also has heavily modified banks (Van Looy et al. 2005). Other research also revealed congruence in diversity patterns for stream biota, with deviations due to variable responses to environmental changes and pressures between different taxonomic groups (Heino et al., 2005).

In conclusion, we can say that for this river, changes in the physical environment and habitat heterogeneity, together with contact to different ecoregions, results in changes in community composition and biodiversity. This happens by the influx of species from the drainage network and immediate surroundings, plus the loss of species that are less well adapted to the altered environmental conditions.

Diversity in composition and richness explained by classifications

In the 2-dimensional DCA-plot (Figure 2.9), the reaches and classifications show some distinctions, but also overlap in places. Axis 2 clearly separates out the downstream longitudinal order of the reaches and is termed the 'order axis'. This downstream organisation in the reaches is mirrored by changes in the adjacent ecoregions, as represented by ellipses and Arabic numbers (Figure 2.9). In upstream reaches, the most specialized functional adaptations are found, with avoiders and resisters prevailing; and moving downstream a gradual trend towards invaders and endurers can be seen.

Axis 1 shows a trend from true floodplain meadow species with adaptations to the river's flooding regime on the right, to invader species with broader, non river-specific habitat requirements on the left. The population strategies show the same trend from disturbance strategies to the left of the axis to the most stable population strategies on the right. As species adaptations and population dynamics show very strong correspondence with this axis, we have termed it the 'disorder' axis.
The free flowing midreach of the Common Meuse, which is coincident with a peak in species diversity, lies at the left extreme of this axis, opposite to the outer upstream and downstream reaches. The outer reaches show the strongest resemblance according to this axis, as was also reflected in the similarity between these reaches. In terms of functional adaptations, the resisters and endurers prefer the less-perturbed reaches, whereas the invaders prefer disorder (Figure 2.9). The species amplitude also corresponded significantly to this axis, as the higher amplitude species are related to a higher disorder.

The proposed classifications explained the diversity patterns very successfully, each providing additional information. The functional adaptations grouping is most indicative of soil conditions, disturbance regimes and management (Lavorel et al., 1997; Pillar, 1999). The classification revealed the longitudinal disorder in the river system, with the ecoregion classification showing the functionality of lateral connectivity in the river system.
The amplitude classification is especially related to disturbance; the generalists (species with higher amplitude) are related to a higher level of disorder. The influx of species from adjacent ecoregions will understandably be more pronounced in

generalist species, which are mostly highly mobile, and opportunists in the floodplain habitats. In the same way, the higher disturbance regimes will also favour the generalist species that are less habitat-selective and benefit from open ground newly generated by the frequent physical disturbance. The disturbance in the river reaches results in larger numbers of higher amplitude species in reaches with high dynamism, in contrast to the less dynamic reaches with low heterogeneity (Lorraine and Tidal Meuse) and higher numbers of specialists (Naiman et al. 1993; Tabacchi et al., 1996; Mouw & Alaback, 2003). Influx from adjacent ecoregions is highest in the high disorder reaches, mainly due to the use of various population dynamic strategies, allowing for high biotic diversity. The population dynamic strategies, identified in the regional scale species abundance and persistence, give the best interpretation of the regional character of disorder. This classification showed strongest correspondence to the compositional aspects of the floodplain meadow vegetation, the result of the selectivity of species to landscape dynamics at the regional level, such as fragmentation and spatial and temporal disturbance patterns (Eriksson, 1996; Hanski & Gilpin, 1997).
Therefore, the Freckleton & Watkinson (2002) population dynamics classification is very useful in terms of the development of conservation strategies (Freckleton & Watkinson, 2003; Jäkäläniemi et al., 2005). It differentiates between spatial scales of population structure, enabling conclusions relating to the necessary aspects of river continuity and connectivity. We retain this classification in our application of the disorder approach to conservation strategies.

A disorder approach to river basins

We believe that catchment disorder is an important element of river systems that is neglected in most river concepts, which tend to seek generalities over river systems and focus on continuity or serial discontinuity along the longitudinal axis. Trying to identify common ground between water courses, discontinuities and irregularities in the catchment are neglected as a continual source of diversity in a physical as well as a biotic sense. As the contribution of subcatchments and (bio)geographical regions accounts for energy and material exchange and influx of organisms and propagules, the configuration of the drainage network (see Benda's network hypothesis) and presence of different ecoregions in it has important consequences for biodiversity conservation and restoration.

The disorder approach brings us back to the river's equilibrium concept, with biodiversity corresponding to the physical characteristics in structural and functional community characteristics (Statzner & Higler, 1985). As the dissipation of energy is spread over reaches, so is the impact of discontinuities and the community responses. Therefore, influences from the surroundings and local disturbances will be adjusted over a reach. Strong differences between reaches can exist due to the river's tendency to minimize the impact of altered conditions at reach scale, so reaches show uniformity in reaction to alterations or modifications. This tendency has important consequences for nature conservation strategies, allowing choices for conservation or restoration to be made at reach level.

When the population structure of the species at risk are known, the implications of this work can be used to enhance conservation schemes (Hansen et al., 1999; Van Treuren et al., 1993; Selinger-Looten et al., 1999; Brys et al., 2003). The population dynamic strategies, explaining the regional persistence and patterns in populations, can be useful for guiding the delineation of conservation strategies (Miles, 1979; Lowe, 2002; Freckleton & Watkinson, 2003). In reaches with mainly patchy regional ensembles and local extended populations, strategies have to be focused on habitat conservation and habitat restoration with emphasis on preservation of populations. For the resilience character of these populations, the availability of suitable habitat prevails and populations are sensitive to factors that change suitability of habitat. Remnant populations are also sensitive to aspects of habitat quality, but a regular generation of new habitat patches is also essential. When metapopulation, source sink and shifting cloud strategies are more abundant or characteristic of the species at risk, conservation strategies need to focus on the processes of habitat creation and recolonization. Measures need to be aimed at restoring disturbance regimes. From our conclusions, the potential for restoration is maximal in high disorder reaches with emphasis on restoring river dynamics. More careful actions, with emphasis on preservation of populations, must be utilized in reaches with lower disorder characteristics.

Conclusion

To conclude, we refer to the general remark of Sabo et al. (2005) that river networks dissect landscapes and provide a natural framework for conservation planning. In our observations, we found proof of the contact between the river and dissected and connected landscapes. The patterns of connectivity and disturbance over the river reaches play a determining role for the diversity of floodplain communities. With the disorder approach introduced here, we revealed the weaknesses in the two hypotheses and some shortcomings in the conceptual frameworks of river ecology. The continuity principle of the River Continuum Concept is too limited a concept for choices in conservation strategies in the riparian zone, as it only envisages continuous patterns and processes in the river. The determined disorder in river communities has stronger predictive power for biodiversity patterns and composition changes between river reaches. It is generated by discontinuities in the physical environment of the river system, caused by discontinuities in the catchment of geomorphic or geographic origin. The disorder aspects in the river corridor can be identified in the biogeographic and functional traits of plant communities. Therefore, we propose them as guiding principles in biodiversity conservation strategies.

The catchment disorder analysis introduced here is an interesting approach in the assessment of restoration and conservation potentials. The connectivity along the river as well as lateral to the river (with adjacent ecoregions) proves significant in sustaining local biodiversity for reaches that show responses to disturbance and environmental gradients. Reaches characterised by lower levels of disorder are more independent of upstream energy, material and propagules. These reaches can be treated as isolated from other reaches and hinterlands, and plans can be elaborated on a more local basis. For high disorder reaches, the contact with adjacent reaches and regions does matter.

Acknowledgements

This collaboration was made possible by the IRMA/SPONGE programme, governed by NCR (National Center for River Research), under commission of the IRMA International Rhine-Meuse Activities sponsored by the European Commission. Flow rate data from the gauging stations was provided by the river authorities in the three countries. Special thanks to Mark Hulme for tidying up the English, and to Jon Sadler and two anonymous reviewers for their highly supporting comments.

Appendix S1 Species matrix for plant data collected in the Meuse sampling, with abundances over reaches and functional group classification.

III

Drivers of riparian biodiversity

In this chapter, the main drivers of biodiversity are identified for the riparian zone of the River Meuse. These can be steering processes or key features providing crucial information on structural and functional components of the river system in the light of our study. Key ecological elements are determined for the different compartments of the river system; for the riverbank zone as well as for the floodplain area, from the larger habitat patches analysis at reach level, to the microhabitat level at site level.

The surveyed scale levels for this chapter are the River Meuse and its middle reach, the Common Meuse reach, in combination with the local site level.

Different processes and features are addressed at the different scales, from flooding, flood interruption and habitat fragmentation and isolation, to local soil, succession, dispersal and recruitment processes.

Central questions are for the steering processes and related problems and threats for the identified key ecological elements, here described at community level. Further question is for the role played by fluvial processes in the habitat conditions.

Themes and groups emphasized upon are riparian and floodplain forests for the larger structures and functions at landscape scale, ground beetles for the small-scale processes and key factors in the riparian zone and dry river grasslands in the floodplain.

III.1

The effects of river embankment and forest fragmentation on the plant species richness and composition of floodplain forests in the Meuse valley, Belgium.

Van Looy, K., Honnay, O., Bossuyt, B. & Hermy, M. 2003. The effects of river embankment and forest fragmentation on the plant species richness and composition of floodplain forests in the Meuse valley, Belgium. Belg. Journ. Bot. 136 (2): 97-108.

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Abstract

We studied the effect of disruption of alluvial forests from natural river flooding on their vascular plant diversity in the river Meuse floodplain in Belgium. The river Meuse is a rain-fed river, originating at an altitude of 409 m above sea level and discharging into the North Sea some 900 km further downstream. The Meuse river was channelized for most of its course in Belgium and The Netherlands during the last 2 centuries. In the study area a continuous embankment was gradually realised with a system of winter dykes. This enabled us to sample forests along a gradient of isolation from the river and flooding frequency. Flooding frequency was the most important correlate of community composition of the forests. Forests still under influence of the river were significantly richer in river species and significantly poorer in woody species than forests disconnected from the river. They also had a higher beta species diversity and tended to have a higher alpha diversity. Disconnected forests seem to loose species but they do not gain species at the same rate. We suggest that the two most important ecological processes behind this are 1) the poor colonisation capacity of typical forest plant species which is mainly due to dispersal limitation and 2) the absence of natural disturbance events which stimulates the dominance of certain competitive species in these forests. Only the re-establishment of lateral river connectivity and natural dynamics can stop this process.

Introduction

Due to their extremely high spatio-temporal habitat heterogeneity and habitat connectivity, river systems belong to the most species rich ecosystems in temperate regions (Nilsson et al. 1989, Gregory et al. 1991, Zwick 1992, Pollock et al. 1998, Ward 1998, Ward et al. 1999, Tockner et al. 1999). Schnitzler (1996) refers to river ecosystems as natural non-steady state systems characterized by short-term cyclic changes. Research on biodiversity gradients within river systems can be categorized into studies of longitudinal diversity patterns (i.e. along the river course) and lateral diversity patterns (i.e. perpendicular to the river course) (Ward 1998). Most research until now dealt with along-stream changes in biodiversity and used the River Continuum Concept as a useful theoretical framework for hypothesis testing, especially for North American rivers (Vannote et al. 1980, and see e.g., Statzner & Higler 1985, Oberdorff et al. 1993, Grubaugh et al. 1996). Lateral biodiversity gradients in the floodplain of natural flowing rivers have also been documented (e.g., Ward & Stanford 1995, Tockner et al. 1999, Lyon & Sagers 1998).

During the last century, almost all large European rivers have been subjected to hydraulic management in order to facilitate navigation and to control flooding. Hydraulic management may affect both lateral and longitudinal biodiversity gradients (Bravard et al. 1986). Dam construction e.g., will mainly affect longitudinal connectivity and hampers the migration and dispersal of species (e.g., Englund et al. 1997, Andersson et al. 2000, Jager et al. 2001) although it may also affect indirectly lateral biodiversity gradients by preventing downstream floodplain flooding (Nilsson et al. 1997). River channelization and dyke construction on the other side mainly affect lateral biodiversity gradients by disrupting the connection of the river with its floodplain. Much less research documenting this process is available so far, as it is only recently that aquatic ecologists appreciated the extent of it (Ward 1998).

Here, we deal with the effects of river embankment on lateral plant diversity gradients in the river's floodplain. In unmodified river systems, the riparian vegetation is expected to exhibit a zonation from the river channel to the uplands along an elevation gradient (Lyon & Sagers 1998, Siebel & Bouwma 1998). In the lower parts of the floodplain, regular flooding creates disturbance and opens space for recruitment of plants that are transported by the river. Along the upward elevation gradient, disturbance and connectivity with the river channel gradually decrease resulting in a characteristic plant diversity gradient. Some authors have compared the plant diversity of riparian zones with the surrounding area free from any river influence and concluded that diversity may be more than two times higher in the floodplain zone (Brown & Lugo 1982, Gregory et al. 1991). Hence it can be expected that disrupting floodplains from their river channel may have a serious impact on biodiversity.

Some attention has already been paid to the effects of flood disruption on species richness and species composition of floodplain forests (Trémolières et al. 1998, Deiller et al. 2001). These authors concluded that forests in the former floodplain of the Rhine River lost their alluvial specificity but did not suffer from a loss in species richness. These studies, however, lacked a statistical comparison of species richness between locations subjected to different flooding regimes and only dealt with woody plants and fungi.

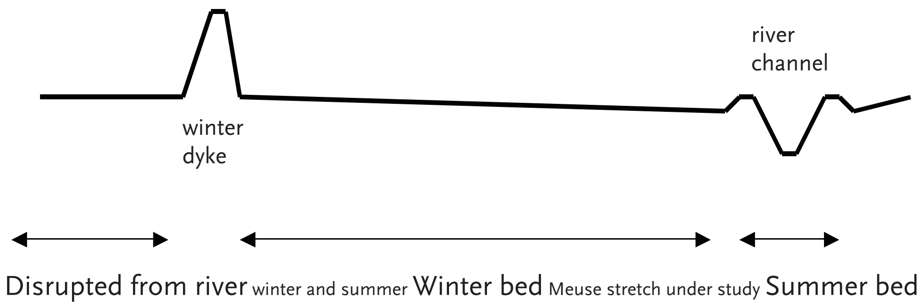


Figure 3.1. System of winter and summer dikes in the border Meuse stretch under study.

We studied the total plant species richness of forests in the floodplain of the river Meuse in Eastern Belgium. The Meuse river was channelized for most of its course in Belgium and The Netherlands during the last 2 centuries. For the study area of the border Meuse stretch with The Netherlands, a continuous embankment was gradually realised with a system of winter dykes (Figure 3.1). This enabled us to sample forests along a gradient of isolation from the river and flooding frequency. The main questions were:

What is the effect of complete disruption of the forests from river flooding on their plant diversity compared to the diversity of forests in the winter bed, which are still under river influence?
Can the plant species diversity and species composition gradient in the floodplain forests (if present) be explained by their flooding frequency?

Material and Methods

Communities, sampling and species diversity

Floodplain woodland communities in the study area show considerable variation but can be divided in three groups. The first community is the most frequent one and may be considered as a willow woodland with Salix alba and S. viminalis as the most common woody species. But also other woody species may occur (e.g. Fraxinus excelsior, Crataegus monogyna, C. laevigata, Alnus glutinosa). Syntaxonomically this community relates to the Salicion albae and at least part of the plots may be described as the Artemisio-Salicetum albae (cf. Hommel et al. 1999:167).
Alnus glutinosa dominates the second community. It is differentiated by a considerable number of Alnion glutinosae-species, e.g. Lycopus europaeus, Solanum dulcamara, Iris pseudacorus. It therefore belongs to the Alnion glutinosae and probably to the Carici elongatae-Alnetum (cf. Stortelder et al. 1999).
The last community is dominated by hard wood species, mainly Fraxinus excelsior, but also Quercus robur and Acer pseudoplatanus occur regularly. It clearly belongs to the Alno-Padion, but typical forest plant species often are absent (e.g. Lamium galeobdolon, Adoxa moschatellina, Primula elatior, Paris quadrifolia).
In general all observed communities are relatively heterogeneous and usually not fully developed. This clearly relates to the highly dynamic nature of these habitats with regular disturbances (flooding, erosion and sedimentation).

55 forest fragments were sampled using 69 plots of 3m*3m size. The forests range from hardwood to softwood forests (although it is not always possible to attribute them unequivocally to one of these categories, and understorey layers between these forests do not really differ). 4 samples (all softwood) were lying in the sum-

merbed, 48 samples (29 softwood and 19 hardwood) were lying in the winterbed and 17 samples (5 softwood and 12 hardwood) were lying in forests completely disrupted from any river influence. Disruption occurred 20-30 years ago for 9 of these disconnected forests and 160-180 years ago for 8 of these forests. Flooding frequency of the samples ranges from more than once a year to less than once within a decade. Flooding frequency was summarized in three categories: 'never to less than once in a decade', 'more or equal to once a year' and 'between once a year and once in a decade'. We also determined for each plot the distance in bird's eye view to the river channel (m). Also soil texture in each sample plot was manually analysed (in 9 ranks, from gravel (9) over clay-sand (5) to clay (1)).

All taxa of vascular plants within the sample plots were recorded using the Tansley scale (Tansley 1935) between 1996 and 1999. The full data matrix is available from the authors upon request. Species were attributed to different functional ecological groups. We distinguished between woody species, typical forest species (sensu Honnay et al. 1999) and typical river species (sensu Malanson 1993 and Berten & Leten 1995). We also calculated the Shanon-Wiener diversity index and a dominance index for each sample plot using the EcoSim software (Gotelli & Entsminger 2001). The Shannon Wiener diversity index includes both species richness and evenness in one index, while the dominance index simply reflects the fraction of the sample plot represented by the most abundant species (Magurran 1988). Finally, beta species diversities of the winterbed and the isolated floodplain zone were calculated together with the beta diversities of the sample plots grouped by flooding frequency. Beta diversity was defined as gamma diversity divided by alpha diversity (Schluter & Ricklefs 1993). Gamma diversity is the total species richness of the group of sample plots under consideration. Alpha diversity is the mean number of species in this group of sample plots. We report the results as the reciproque of beta diversity as this measure expresses the mean number of sample plots where a species occurs. We used a Jack-knife method (Manly 1997) to generate a variance for each calculated beta diversity value in order to be able to compare the beta diversity between groups of sample plots.

Analyses

In a first stage we performed a data exploratory Detrended Correspondence Analysis (DCA) using the CANOCO 4.0 software (Hill & Gauch 1980, ter Braak & Smilauer 1997). In order to identify the abiotic drivers of the species composition gradients, DCA sample scores of the 69 sample plots were related with flooding frequency and river channel connectivity (summerbed, winterbed or disrupted from the river) using a one way Analysis of Variance (ANOVA) and with distance to river and soil texture using a Spearman rank correlation coefficient. DCA sample scores were normally distributed (Kolmogorov-Smirnov-test) and exhibited equal variances across examined class variables.
Next, to investigate whether the construction of winter dykes and the associated disruption of forests from the river affected the species diversity of these forests we compared their community composition and species richness with the forests situated in the winterbed using an independent two sample t-test tests. Because there are only 4 summer bed samples these were omitted from this analysis.

Finally we aimed at revealing the ecological mechanism behind the differences in species diversity and we related species richness and diversity of the sample plots with their flooding frequency. We used a one-way ANOVA with Tukey pairwise comparisons between the three levels of flooding frequency. All statistical analyses were performed with SPSS 10.0.5. The jack-knife procedure was performed using the S-Plus software.

Results

We found a total of 209 plant species. Species and the species subgroups are listed in the appendix table S2. Species are ranked according to their score on the first DCA axis (representing increasing flooding and disturbance). The first two axes of the DCA only explained a cumulative percentage of the variance of the species data of 10%, expressing the very heterogeneous character of the vegetation in the sample plots (Figure 3.2). There was a significant relation between DCA1 sample scores and flooding frequency and river connectivity. DCA2 sample scores were significantly correlated with flooding frequency and with soil texture. We found no correlations with distance to the river (Table 3.1, Figure 3.3).

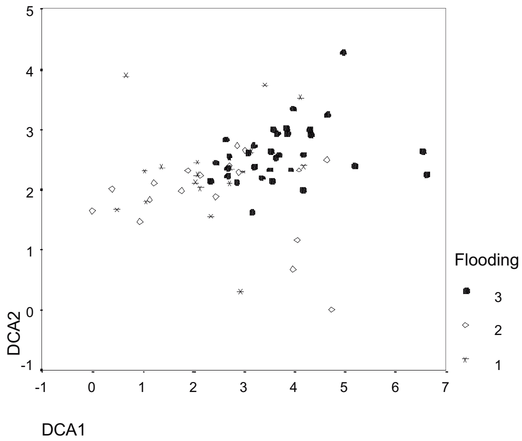


Figure 3.2 DCA sample scores of the sample plots marked by their flooding frequency. 1: never to less than once in a decade; 2: between once a year and once in a decade; 3: more or equal to once a year.

Table 3.1 Relation between the DCA sample plot scores and three variables measuring river dynamics in each sample plot (n=69).

	Flooding frequency[1]	River connectivity[1]	Soil texture[2]	Distance river2
DCA1	11.54***	11.48***	-0.16	-0.05
DCA2	6.97**	0.04	-0.38**	0.15

[1]F-values (ANOVA, k = 3), [2]Spearman rank correlation coefficient,

*level of significance:*** : p £ 0.001; ** : 0.001 < p £ 0.01 (2-tailed p-values)*

Table 3.2 Average species number and species diversity of the sample plots in the winter bed and in the zone which is completely disrupted from river influence. Comparison of means using an independent sample t-test (n=65).

	Disconnected	Winterbed	t-value
All species	15.4	17.4	1.14
Forest plant species	10.4	9.4	0.78
Woody species	4.9	3.9	1.75(*)
River species	1.4	3.2	3.30**
Dominance	0.22	0.18	1.78(*)
Shannon Diversity	2.27	2.43	1.44
1/Beta diversity (samples/species)	4.64	2.49	1.99*

*** : 0.001 < p £ 0.01; *: 0.01 < p £ 0.05; (*) : 0.05 < p £ 0.1 (2-tailed p-values)*

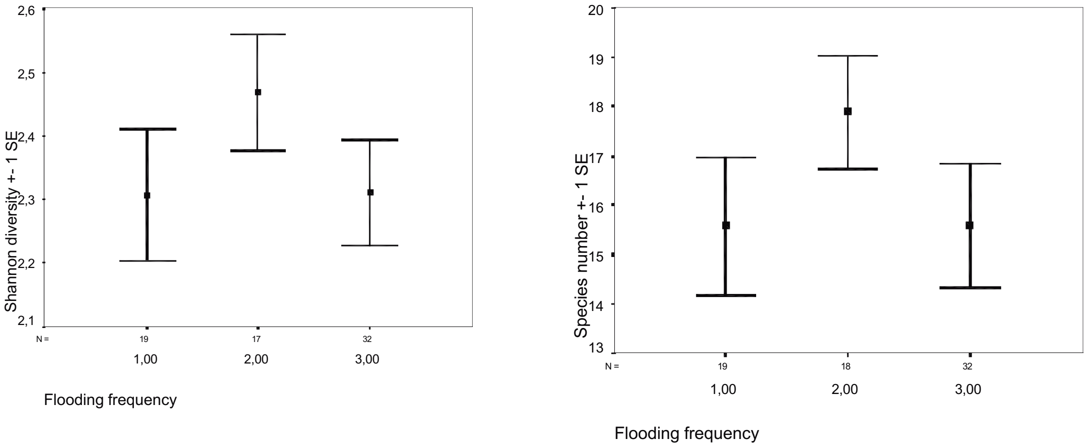


Figure 3.3a and b. Relation between species richness (a) and the Shannon diversity index (b) and the flooding frequency (1: never to less than once in a decade; 2: between once a year and once in a decade; 3: more or equal to once a year).

Forests in the winter bed were significantly richer in river species and marginally significantly poorer in woody species than forests disconnected from the river (Table 3.2). This resulted in a tendency for winter bed forests to be more species rich than disconnected forests, although the difference is not significant. Disconnected forests were also more homogeneous in species composition: they had a higher dominance value and a lower beta diversity (i.e. species are, on average, occurring in more sample plots than in the winter bed forests).

Frequently flooded forests contained marginally significantly less forest plant species, less woody species and significantly more river species (Table 3.3). This resulted in maximal species richness and species diversity in the forests with an intermediate flooding regime, although the differences with the higher and lower flooding regime forests were not significant (figure 3.3a and b). Beta diversity is significantly higher in the forests with the intermediate flooding regime (Table 3.3).

Table 3.3 Effects of flooding frequency on average species richness and diversity (n=69). Different letters represent significant differences (p<0.1) according to Tukey pairwise comparisons.

	1 (<=once /10y)	2 (2-9 /10y)	3 (>=once /y)	Overall F value
All species	15.6	17.9	15.8	0.81
Forest plant species	10.2ab	10.6a	7.8b	2.81(*)
Woody species	4.8b	4.2ab	3.4a	2.65(*)
River species	1.4a	3.0b	3.4b	6.17**
Dominance	0.21	0.19	0.21	0.58
Shannon Diversity	2.31	2.47	2.31	0.81
1/Beta diversity (samples/species)	3.61a	2.40b	3.55a	4.36*

*** : 0.001 < p £ 0.01; *: 0.01 < p £ 0.05; (*) : 0.05 < p £ 0.1   (2-tailed p-values)*

Discussion

As appears from the DCA result, flooding frequency is the major driving force behind the plant community composition of forests in the Meuse river floodplain. Other variables like river connectivity, distance to river channel and soil texture are less effective in explaining community composition. Soil texture is a secondary and complex variable that is the result of the interaction between flooding frequency and the presence, dimension and shape of geomorpholoical features in the landscape (Bornette & Amoros 1996). River connectivity and bird's eye distance to the river channel are moderate to very poor correlates of flooding frequency, which is also due to the geomorphological heterogeneity of the floodplain.

Forests in the winter bed tend to be more species rich than forests disconnected from the river although the difference is not statistically significant. River species in particular disappear from the forests when they are disconnected from river influence. Most of these river species are dependent on free space generating disturbance events for their germination and recruitment on the one side and on flowing water for their dispersal on the other side (Bornette & Amoros 1996, Ward et al. 1999). Both processes (i.e. recurrent disturbance and habitat connectivity through flowing water) are lacking in the forests that became disconnected from river influence. The importance of disturbance events in structuring the plant communities is confirmed also by the (on average) higher dominance index and the lower beta diversity of the disconnected forests. In the absence of disturbance some plant species tend to become dominant on the sample plot scale and between sample plots. This results in a higher dominance of certain species and in the relatively high number of sample plots were individual species occur (i.e. in a low beta diversity). Regular disturbance events in the winter bed due to flooding leaves less opportunities for certain plant species to become dominant. It is known that especially in highly productive ecosystems like most river systems, only when some species are eliminated regularly, species diversity can be maintained. This corresponds to Huston's so called general hypothesis of species diversity (Huston 1979, Ward et al. 1999). Generally, flood-created disturbances provoke rejuvenation of floodplain zones with patches of different degrees of maturity, resulting in a meta-stability of plant and animal communities (Bretschko 1995).

This conclusion is also in accordance to the homeorhesis (from the Greek "preserving the flow") concept as an important issue in dynamic systems. The analysis of the system in motion, incorporating the processes and the meta-stability of the dynamic system, is the challenge for the planning of the restoration schemes. The ecological restoration focusses on enabling the river dynamic processes that maintain the floodplain habitat heterogeneity. For the restoration, i.e. regeneration of floodplain woodlands, flooding events are documented as an essential feature (Schnitzler 1997).

Besides their loss of river species, disconnected forests tend to become enriched with forest plant species, although the latter is not as explicit as the first. The habitat of the disconnected forests slowly changes and becomes more and more suitable for the establishment of typical forest plant species. Three interconnected processes may be responsible for this process:

1. Lowering of the ground water table,
2. Gradual colonisation of understorey woody shrub species which decrease light transmission to the soil and hence the cover of very competitive tall herb species (Siebel & Bouwma 1998, Deiller et al. 2001). It is known from other forest ecosystems that tall species like Urtica dioica and Rubus fruticosus

coll. may prohibit forest plant species colonisation (Honnay et al. 1999b, Verheyen and Hermy 2001),

3. Decrease in soil phosphate level which also has a negative impact on the presence of the tall herb vegetation and hence a positive on forest plant species colonisation (Trémolières et al. 1998, Honnay et al. 1999b). The biodiversity lost by the disappearance of river species, however, does not seem to be completely compensated by enrichment in typical forest species. Most forest plants species have no special seed adaptations for long distance dispersal (Hermy et al. 1999, Butaye et al., 2001). Hence it is very difficult for these species to reach these fragmented forest patches although they seem suitable for colonisation after the termination of the flooding events. In other words: the loss of river species is hardly compensated by the colonisation of typical forest plant species due to the fragmentation of the forests. In the winter bed, forest fragmentation is no problem as the connectivity between forest species is guaranteed due to regular flooding events. The impoverishment process of the disconnected forests is confirmed when the recently (20-30 yrs) and long time ago (160-180 yrs) disconnected forests are compared (results not shown). The latter contain significantly less river species than the former and only marginally signifi cantly more forest plant species.

Trémolières et al. (1998) and Deiller et al. (2001), in contrast, found an accumulation of species in Rhine alluvial hardwood forests along a gradient going from still flooded, unflooded for 30 years and unflooded for 130 years. These authors, however, only studied woody plants and found an accumulation of species like e.g. Viburnum spp., Ligustrum vulgare, Prunus spinosa, Berberis vulgaris, Acer campestre and A. pseudoplatanus. These tree and shrub species are ornithochoreous or anemochoreous and hence are dispersed relatively easy over long distances. The forests that we studied became also significantly richer in woody species after disconnection from the river. However, the dispersal and establishment process of most herbaceous forest plant species seems much more problematic.

In order to get insight in the basic ecological process responsible for differences in species richness between disconnected and winter bed forests, we also directly focused on the effects of flooding frequency on species richness. As already mentioned when discussing the DCA results, flooding frequency is a more precise exploratory variable than river connectivity because even in one connectivity class (the winter bed) there is a flooding frequency gradient which is associated with the presence of large geomorphological features, rather than with the distance to the river channel (cf. Table 3.1). Species richness and species diversity (the latter expressed as Shannon diversity) tend to peak at intermediate flooding frequency (figure 3.3a and b). At low flooding frequency, certain species tend to become dominant as already discussed in a previous paragraph. When flooding frequency becomes too high, only a small number of specialist pioneer species can survive. This is completely in accordance with the intermediate disturbance hypothesis which again proofs to be a very powerful theoretical framework (Connell 1978, Huston 1979, Bartha 1997). Ward et al. (1999) recently confirmed the relevance of general biodiversity concepts derived from terrestrial and marine environments to river systems.

Conclusion

Forests disconnected from river influence loose typical river species and do not gain forest plant species at the same rate. Disconnected forests tend to become less species rich and have a lower beta diversity due to the increasing dominance of certain plant species. The ecological rationale behind this species loss is 1) dispersal limitation of typical forest plant species in reaching the fragmented disconnected forests and 2) the intermediate disturbance hypothesis that predicts a decrease in species richness in the absence of disturbance events prohibiting dominance of certain competitive species.

III.2

The role of river dynamics in the conservation of dry river grasslands

Kris Van Looy

Submitted to Journal of Applied Vegetation Science

Abstract

Question

In what way and to what extent do river dynamics determine the development, conservation and restoration of dry river grasslands?

Location

The River Meuse is one of the larger Northwest European streams with nature protection that places emphasis on river corridor plants in dry river grasslands. The Common Meuse reach is a 30km unregulated river stretch at the border between Belgium and the Netherlands.

Methods

The grasslands of the alluvial plain were mapped and sampled with vegetation relevees and soil sampling. Spatial information gathered using GIS was added to this data matrix, and a hydraulic model added river variables of flood frequency and flow velocity. Ordination and diversity analysis was carried out to link composition and diversity aspects to soil conditions and river dynamics, the results of which were used to build a community distribution model. As the dry river grassland is a threatened vegetation type, an analysis of species at risk was performed to find key constraints and define effective restoration measures.

Results

Flood dynamics, soil conditions and management proved to be determining aspects for the composition and diversity of river grassland. The different dry river grassland communities were clearly distinguished by soil pH and salt concentration – soil variables that were significantly correlated to the flood regime. The group of river corridor plant species studied were good indicators of well developed dry river grassland patches. The significant isolation aspect of the dry river grassland relicts was found to be due to recruitment limitation, as a consequence of habitat fragmentation linked to land use intensification and river regulation. As the habitat creation process is the trigger for sustainable conservation, a spatial model based on hydraulic modelling using GIS allowed the prediction of potential dry river grassland development and gave insight into spatial and management requirements for conservation strategies.

Conclusions

Knowledge of habitat conditions and dynamics is essential when forming conservation strategies for dry river grasslands. The river corridor plants proved to be a good flagship species group for the protection and restoration effort.

Introduction

Conservation and restoration of river landscapes receives much attention today, especially hot spots for diversity within the river corridor, such as the dry river grasslands (Jongman 1992, Stroh et al. 2005). In Northwest Europe's large river valleys, characteristic landscape features are levees and dikes in the floodplain, which can lead to the development of dry river grasslands. A group of river corridor plants reached these habitats by moving along the large floodplain corridors of these rivers. The natural or artificial levees are exposed to the sun and warm up quickly, creating favourable conditions for species that would normally occupy a distribution range more to the southwest; the species of the calcareous upstream regions in particular take advantage of these opportunities. River regulation, intensification of agricultural practices and disconnection of parts of the floodplain area are the main pressures for the river forelands of the large Northwest European rivers. These factors pose a huge threat for river corridor plants through increased habitat fragmentation (Burkart 2001, Donath et al. 2003, Wolfert et al. 2002).

A main focus for this research was the role that river dynamics could play in the potential rehabilitation of dry river grasslands. This central theme could be further divided into the following questions:

1. What physical variables are the most important for determining the composition and diversity aspects of these grasslands?
2. What parameters are of importance for the conservation and regeneration of habitat in space and time?
3. How can these variables be governed/controlled?

For the first question, we sampled grasslands over the Common Meuse valley and tried to identify, using ordination techniques, the determining factors for composition and richness of the vegetation, bearing in mind the special emphasis that must be placed on soil characteristics related to geomorphological processes when developing spatial predictive models for riparian vegetation (Toner & Keddy 1997, Richards et al. 2002). For delineating and choosing conservation and restoration options, we selected the rare species and the habitat at risk, the dry river grasslands being rich in river corridor species; and carried out a diversity analysis to address whether the river corridor species are a good indicator group, and so whether they can function as flagship species.

Regarding the second and third questions, river concepts and modelling approaches can be useful for defining and solving the problem. When searching for effective conservation strategies, river concepts such as the 'shifting mosaics' and 'patch dynamics concept' provide useful frameworks for the definition of spatial and management guidelines (Pickett & White 1985, Petts & Bradley 1997), and river dynamics play a central role in shaping the river landscape according to these concepts. Many authors have described flooding as the driving force behind the composition and diversity of floodplain vegetation (Heiler et al. 1995, Tabacchi et al. 1996, Tockner et al. 1999), and yet several other authors have stressed that the flood intolerance of dry river grasslands is a central conservation aspect (Jongman 1992, Vervuren et al. 2003, Eck et al. 2005). By analysing the determining factors, this study attempts to develop a model to predict the potential recovery of these threatened communities, and to derive guidelines from present distribution patterns for their conservation and restoration. These guidelines will then be useful for further planning of the restoration programme for the river's floodplain.

Study area

The site used for this study was the alluvial plain of the middle course section of the River Meuse between Maastricht and Maaseik on the border between Belgium and the Netherlands (reach of 30 km), known as the Common Meuse. The Meuse is a rain-fed river, originating at an altitude of 400 m above sea level at the Plateau of Langres in the North of France and discharging into the North Sea some 900 km further downstream. The discharge of the Meuse shows great fluctuations due to its rain-fed character and the rocky soils of the Ardennes upstream catchment (Pedroli & De Leeuw 1997), ranging from 10 m?/s during dry periods to 3,000 m?/s during periods of heavy rainfall within the catchment area. The unregulated Common Meuse stretch is a typical gravel river with a strong longitudinal gradient (0.45 m/km), the valley consisting of a gravel underground with a loamy alluvial cover. Local irregularities of levees and dikes are covered with more sandy sediments, as are the dynamic overbank sedimentation zones. The floodplain has been traditionally used as meadows for agriculture. Large parts of the alluvial plain have been excavated for gravel mining, leaving large gravel pits or lowered floodplain zones. The degradation of the floodplain natural heritage was the reason for developing a river restoration programme and to start local pilot areas, mostly in abandoned gravel mining locations. The project is defined within a larger master plan for the whole alluvial plain (Pedroli et al. 2002).

Survey and sampling

Valley and dry river grassland sampling

The vegetation survey for the Flemish side of the Meuse alluvial plain consisted of vegetation mapping with sampling for every recorded patch (Figure 3.4). For delineating homogeneous vegetation patches in the field, a minimum of 500 m? was chosen, and the basis for mapping was topographic. The patches of intensive agricultural land use were mostly uniform in terms of vegetation composition, but for the naturally managed areas more irregular patch forms were present. The 196 patches of grassland under natural or extensive management were sampled in 1999 using 1 x 1m relevees, and all species within the sample plots were recorded using the Tansley scale (Tansley, 1935). The grasslands cover a range of types, from open pioneer to dense, tall vegetation. They were classified into nine types according to river dynamics, elevation and management (Table 3.4), and were then assigned to a corresponding phytosociological association or order according to Weeda et al. (1998).

Table 3.4 Classification of grassland patch types in the Meuse alluvial plain, with annotated phytosociological communities (Schaminée et al. 1998).

Agricultural practice		
	B1 hayfields	*Arrhenatherion elatioris*
	B2 pastures	*Cynosurion cristatus*
	B3 fertilised meadows	*Poö-lolietum perenne*
Natural management		
Lower floodplain meadows	F7 long inundated meadows	*Lolio-potentillion anserinae*
	F9 floodplain meadows	*Alopecurion pratensis*
Higher floodplain meadows	L1 dry river grasslands	*Medicagini-avenetum pubescens*
	L4 xeric grasslands of open sand	*Thero-airion caryophyllea*
Overbank sedimentation zones	A1 gravel overbank sedimentation	*Alysso-sedion albi*
	A2 sand overbank sedimentation	*Sedo-thymetum pulegioides*

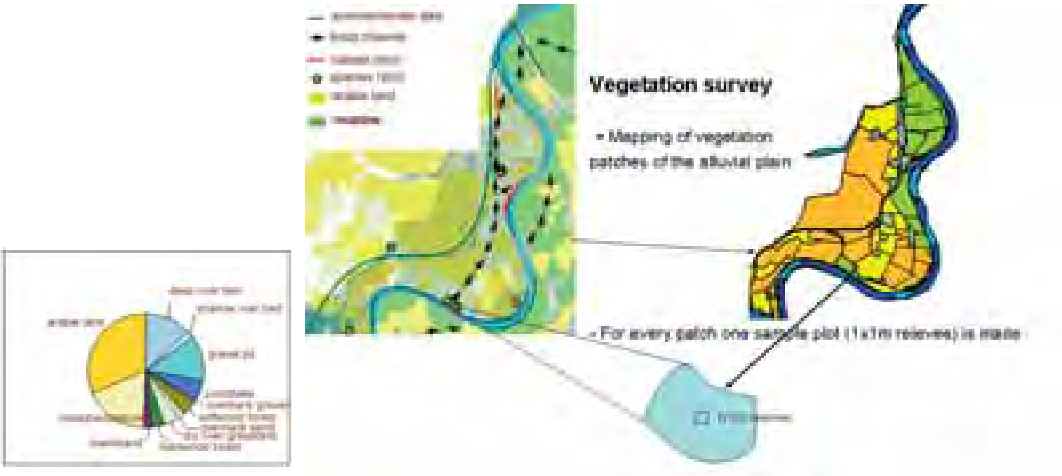


Figure 3.4 Illustration of the alluvial plain survey with an inset of results in a pie chart.

Environmental variables were gathered in the field, or derived from available digital flooding data, and also through GIS mapping. Flooding frequency of the samples ranged from more then once a year to less than once a decade, and was derived from a two-dimensional hydraulic model, developed for the restoration model, and based on a high resolution DEM of the alluvial plain. The frequencies were divided into flood frequency classes (0: >1/year; 1: 1/year; 2: 1/2–5 years; 3: 1/5–10 years; 4: < 1/10 years), and flow velocity (m/s) was also derived from this model, retrieved from the grid cells during a decennial flood episode. In addition, the distance to the river's main channel (m) was determined. Isolation data were ranked into categories, measured as the distance to the nearest same patch type (1: <50m; 2: 50–500m; 3: 500–2000m; 4: >2000m). Management was classified as: extensive meadows with haying and/or pastures (2); natural grazing (1); or no management (0). Soil humidity was classified as: wet (3); periodically wet with high fluctuation (2); moderately dry (1); or extremely dry (0). The organic matter in the topsoil layer was categorized as: a thick humus layer (2); present (1); or absent (0). Soil texture in each sample plot was manually analysed and recorded into nine categories: clay (1); silt (2); loam (3); sandy loam (4); loamy sand (5); clayey sand (6); sand (7); gravel–sand (8); or coarse gravel (9).

A set of 50 topsoil (20cm) samples was taken from the grassland sampling plots, uniformly distributed over the Meuse valley gradients of distance to river and elevation, to specifically look at the soil conditions for grassland species in their root zone. As several authors (Grévilliot et al. 1999, Wolfert et al. 2002, Donath et al. 2003) have identified soil conditions as critical for these communities, emphasis in this study was placed upon these factors in relation to the flooding regime, and relevant soil parameters such as texture, pH, conductivity and organic matter content were measured in the laboratory. Analysis of soil texture was carried out by laser diffraction; pH by Metrohm (titration, pH-carrousel); organic matter by Moffeloven destruction analysis; and conductivity by measuring EC with a conductance meter and translating the readings to soil salt concentration. The surveyed abiotic conditions were screened for correlations using the Pearson correlation test.

Dry river grassland communities and habitat conditions

For the dry river grasslands of the Common Meuse, four vegetation types can be distinguished within the habitat type classification of the European Natura2000 habitat network (EC/92/43, 1992) (Table 3.5). The communities of dry river grasslands can be defined by specific types of pioneer associations of Alysso-Sedion albi (gravel substrates) and the Sedo-Thymetum pulegioides or Thero-Airion caryophyllea of sandy overbank deposits. Succession leads to the dry river grasslands of the Medicagini-Avenetum pubescens or other Koelerio-Corynephoretea communities.

The species of these communities are mostly small and uncompetitive, and for colonization of the habitat to take place, open pioneer conditions are needed. In the xeric Thero-Airion grasslands, two directions of development can be identified: the development of river dune communities, with a richer initial phase; and the continued dynamics of wind and grazing preventing the stabilisation of the stands. Here, the more nutrient-dependent species such as Geranium columbinum and Medicago species indicate an initial phase of development to xeric river dune communities of Thero-airion closer to the Sedo-Cerastion, with the presence of Vulpia bromoides, V. myuros, Carex arenaria and C. hirta. The stands further from the river develop into communities of more acid xeric soils, with the presence of Filago minima and Corynephorus canescens. The most common grasslands of this group are examples under pasture, with a mixture of Thero-airion and Galio-trifolietum grassland, and were classified as an extended Thero-Airion+ community.

Riverine Habitat	Habitat type (EU Habitat Directive)	Characteristic River Meuse corridor species
Overbank gravel deposition A1	6110 pioneer vegetation of calcareous stony substrates Alysso-sedion albi	*Sedum album, S. acre, Poa compressa, Saxifraga tridactylites, Sedum sexangulare, Erophila verna, Galeopsis angustifolia, Geranium pusillum, Kickxia elatine, K. spuria, Lepidium campestre, Verbascum blattaria, Torilis arvensis, Picris echioides*
Overbank sand deposition A2	6120 pioneer grassland of calcareous sandy soils Sedo-thymetum pulegioides	*Sedum reflexum, Avenula pubescens, Carex caryophyllea, Herniaria glabra, Sedum sexangulare, Thymus pulegioides, Potentilla neumannia, Cerastium pumilum, Ononis repens, Potentilla argentea, Valerionella locusta*
Dry river grassland L1	6120 Dry river grasslands of calcareous soils Medicagini-avenetum pubescens	*Medicago falcata, Salvia pratensis, Sanguisorba minor, Plantago media, Scabiosa columbaria, Eryngium campestre, Trifolium striatum, Trifolium scabrum, Anthyllis vulneraria, Rhinanthus alectorolophus, Rhinanthus minor, Vulpia myuros, Trifolium campestre, Leontodon hispidus, Malva mosschata, Tragopogon pratensis*
Xeric grasslands L4	2330 open grasslands of xeric sandy soils and river dunes Thero-airion caryophyllea	*Aira caryophyllea, Carex arenaria, Myosotis ramossisima, Corynephorus canescens, Teesdalia nudicaulis, Ornithopus perpusillus, Filago minima, Medicago arabica, Arenaria serpylifolia, Cerastium glomeratum, Hieracium pilosella, Luzula campestris, Vulpia myuros, Geranium columbinum, Koeleria macrantha.*

Table 3.5 Habitat types of dry river grasslands (including European Natura2000 habitat code) and the associated River Meuse corridor species (italics: diagnostic species following Jansen & Schaminée 2003).

The four types of dry river grasslands (L1, L4, A1, A2) in the valley survey correspond to these vegetation types, and the sampled set of dry river grasslands in our survey were attributed to one of these communities owing to the presence of characteristic species. Thirty-nine well developed dry river grasslands were selected (defined as having > 1 diagnostic species), and for each of these plots sampling of the topsoil was carried out.

Ordination and diversity analysis

A data exploratory Detrended Correspondence Analysis (DCA) using the CANOCO 4.0 software (Gauch 1982; ter Braak & Smilauer 1997) was performed, with only species occurring in more than one plot used for the analysis. A direct gradient analysis for the whole set of environmental variables was run with CCA to reveal relationships between species and the environment. In order to identify the specific contribution of the abiotic drivers of the species composition gradients, the DCA sample scores of the 196 sample plots were related – using a one way Analysis of Variance (ANOVA) – to: flood frequency classes, river channel connectivity, isolation, humidity and vegetation type; and – using a Spearman rank correlation coefficient – to distance to the river and soil texture. This analysis of individual environmental variables was carried out in order to derive relationships useful for a modelling approach.

Guisan et al. (1999) showed that spatial models give better predictions when explanatory variables can be selected, while in CCA-based models and responses, a set of composite environmental variables is applied.

Finally, with the aim of revealing the ecological mechanism behind differences in species diversity, species richness of the plots was related to the environmental parameters using a one way ANOVA, with rare species in the plots being selected and the process repeated. All statistical analyses were performed using the Statistica software package (StatSoft Inc. 2001).

Spatial modelling of dry river grassland community distribution based on hydraulic modelling

A two-dimensional hydraulic model of the Common Meuse reach was developed as part of the river restoration project, using a very detailed DEM basis (1x1m grid, resolution 5cm), and also provided the parameters of flood frequency and peak flow velocity for our mapping units. The stream velocity and shear stress calculated for each grid cell at a given discharge give a reliable measure of the occurrence and delineation of erosion and deposition of gravel and sand (Van Looy et al. 2005). As we determined spatial and flow regime variables for the dry river grasslands, the aim was to integrate these into an expert model to predict the development of these communities over the present and future floodplain, after river restoration. To validate the model results, a field survey was carried out to determine the presence of characteristic species of the Meuse dry river grasslands, and compared to the model predictions.

Alluvial plain grassland survey

Mapping and sampling

The mapping shows that over 50% of the alluvial plain is under intensive agricultural use (Figure 3.4). The dry river grasslands, together with their pioneer stage of overbank gravel and sand depositions, cover only 4% of the alluvial plain, which consists mainly of larger patches of agricultural use (mean area of arable land patches = 4,2ha). Nature reserves and riverbanks account for small vegetation patches.

Analysis of composition and abiotic conditions

The vegetation survey yielded 329 species, of which 226 were present in more than one, and were therefore used for the ordination analysis. For the rare species analysis, 46 species occurring in 2–5 plots (Table S3 in annex) were used. Species richness and patch area were plotted for the grassland types (Figure 3.5), and indicate that the dry river grasslands (types A1, A2, L1 and L4) are the richest communities over the alluvial plain, but with the smallest patches. This indication is even stronger in the species–area plot (Figure 4.10), which shows there is a strong concentration of rare species in the smallest and most species-rich patches. More than half (27 out of 46) of the rare species are river corridor species of the dry river grasslands (Table in annex) of the northern Central European streams (Burkart 2001). Together with the 10 diagnostic species of the dry river grassland associations mentioned above, over two-thirds of the rare species are dry river grassland specialists.

Soil analysis

The measurements taken demonstrate the extremes in soil conditions over the floodplain (Figure 3.6), the conductivity providing an indication of the available salts in the soil, and so the available nutrient content. The extremes over the valley were low pH values (down as far as pH 3.5) for leached higher soils disconnected from the river, and high salt concentration values (up to 316) for frequently flooded silty soils and recent depositions. Most samples yielded high pH values (median of pH 7.25) and low conductivity and salt concentrations (median of 74) due to frequent flooding of, and dry summer conditions for, the floodplain soils respectively.

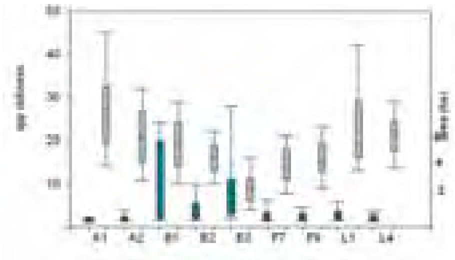


Figure 3.5 Spread of surface and species richness for the grassland patches of the Meuse alluvial plain. Areas in dashed boxes; species richness in grey filled boxes. Bars denote medians and 95th percentiles; boxes denote the 75% confidence intervals.

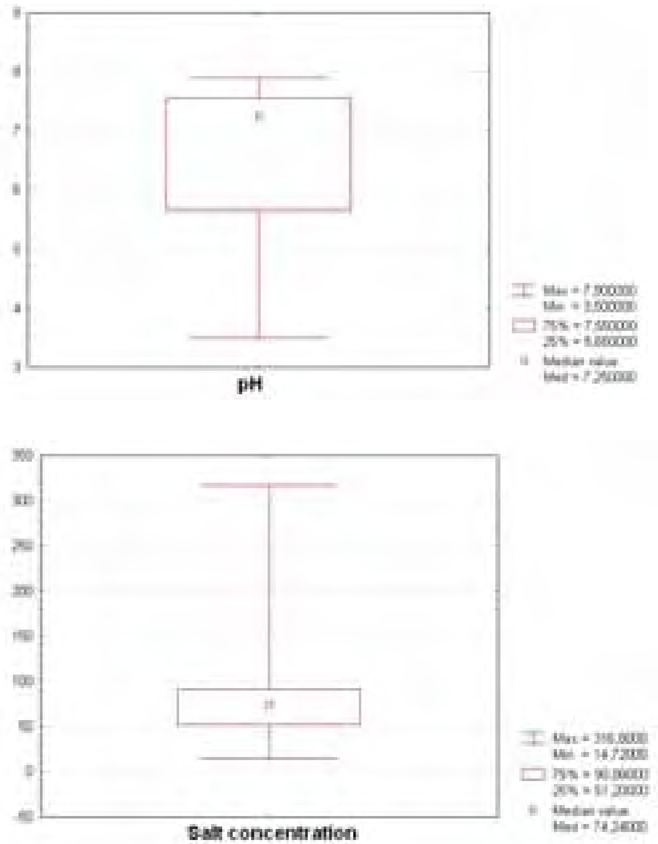


Figure 3.6 pH and salt concentration, as a function of conductivity of the soil (salt concentration=EC (◀S/cm) x 0.64), of topsoils in the Meuse valley (based on a random soil sampling campaign of 80 samples).

Pearson correlation testing revealed significant correlations for conductivity with flood frequency and soil texture; and for pH with flood contact and distance to the river (Table 3.6). Flooding frequency and soil texture also shows a correlation.

Table 3.6 Pearson correlation testing of sampling results for conductivity and pH with spatial variables.

Variable	Conduct.	pH (KCl)	Distance	Frequency
PH (KCl)	-.05			
Distance	.28	-.43*		
Frequency	.46*	.16	.01	
Texture	-.37*	.14	-.17	-.49*

(significant at $p<0.05$)*

The pH of the soil becomes enriched with each flooding event, providing a buffer capacity to soils of the floodplain. Zones further from the river are less buffered because the flood water gradually loses its material and interference with groundwater occurs. Soils are also enriched by salts when flooding occurs – measured by their conductivity – and is particularly pronounced in humid soils and silty sediments. Nutrient availability is therefore related to flood frequency and soil texture and is highest in lower floodplain zones. Higher pH values, however, are more common in higher elevated sandy depositions, closer to the river.

Ordination results

The first three axes of the DCA explained a cumulative percentage of the variance of the species data of 19%, with gradient lengths > 5, expressing the heterogeneous character of the vegetation in the sample plots. For the CCA, the cumulative percentage of explained variance was only 4.9% for the three canonical axes; nevertheless in the Monte Carlo test, eigenvalues and species–environment correlations for the dataset were significant at p:0.005. Inter-set correlations for the seven abiotic conditions showed correlations between soil texture, humidity and flood frequency with the first axis; management and flow velocity with the second axis; and isolation with the third. Dry river grasslands and their diagnostic species are situated in the upper left quarter of the CCA biplot (Figure

3.7) and are strongly correlated to axis 2, indicating a link to river dynamics and management to be a crucial factor for the habitat. Correlation to axis 1 – linked with soil conditions and flooding frequency – is less significant.

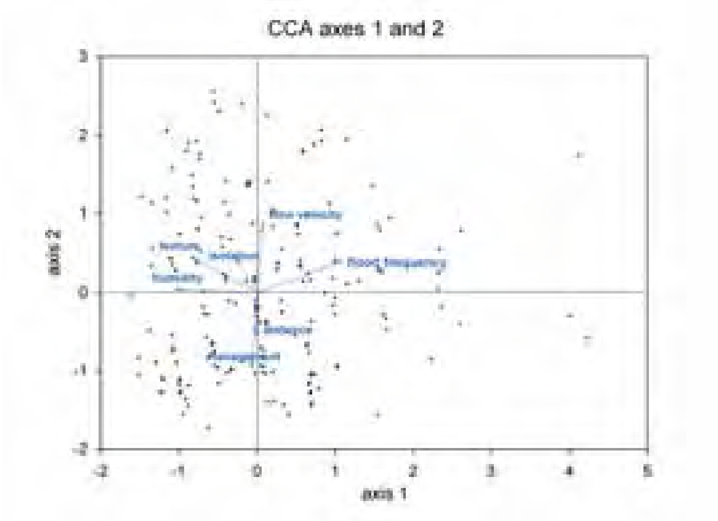


Figure 3.7 Joint plot of the first two CCA axes with plots and environmental parameters.

To reveal further relationships between the abiotic drivers of species composition, the DCA sample scores were analysed against environmental variables. The first DCA axis shows a wet–dry gradient. There are only a few wet meadow patches present and, for the most part, the summer groundwater levels in the alluvial plain are around 3–5 meters below the surface as a consequence of river bed incision during the last century, a process that has been identified for most large European gravel rivers (Bravard et al. 1986, Girel et al. 1997, Piégay et al. 2005). The significant relationship between DCA1 sample scores and flooding frequency, as well as soil characteristics such as texture, soil humidity and organic matter content demonstrates the influence of the river on floodplain environmental conditions. DCA2 sample scores showed significant covariance with soil parameters and management conditions, as well as with isolation and distance to the river (Table 3.7). For this axis, management and soil texture are the most influential abiotic characteristics, showing a gradient of densely vegetated floodplain meadows to open, sandy pioneer grasslands. Hayfield species and nutriphilous species have low values, whereas species preferring sand and calcareous, xerophilic conditions have high DCA2 values. For the third axis, flooding frequency and distance to the river show strong covariance; there is no significant correlation between the dry river grasslands and this axis due to some being close, and others being far, from the river.

Table 3.7 Covariances for environmental variables with ordination axes and species richness.

	distance river	texture	flood freq	contact	isolation	humidity	organic matter	flood class	management
DCA1	0,18*	0,33**	-0,4**	2,95	1,11	37,5**	32,9**	32,6**	4,9*
DCA2	-0,27**	0,45**	0,04	2,15	8,3**	6,7**	17,5**	0,5	10,1**
DCA3	0,38**	0,02	-0,38**	3,12	0,42	0,81	0,74	6,7**	1,2
DCA4	0,2*	0,089	0,07	10,8**	2,3	2,5	0,8	1,2	0,99
Spp richness	0,14	0,16	-0,22*	0,02	2,1	3	5,22*	3,08*	0,9

***red: significant correlation (p< 0,001),*

**green: little significant correlation (0,001< p < 0,01)*

Diversity and rare species analysis

Overall species richness of the samples only shows marginally significant relationships with flooding frequency and organic matter (Table 3.7). For aspects relating to diversity, a clear pattern arises when the rare species are selected; the strong covariance detected between the number of rare species in a plot and the species richness of the plots (F: 3.6, $p<0.001$) shows that these rare species are good indicators for well developed grassland patches. A greater level of rare species is significantly linked to the higher gravel-sandy soils (F:4.6, $p<0.001$), which are the stand conditions of the dry river grasslands.

As the DCA explained most of the variance in species composition, in the diversity analysis covariances between environmental variables and species richness of plots with DCA scores were derived. The rare species were also projected over the two-dimensional space in the DCA biplots.

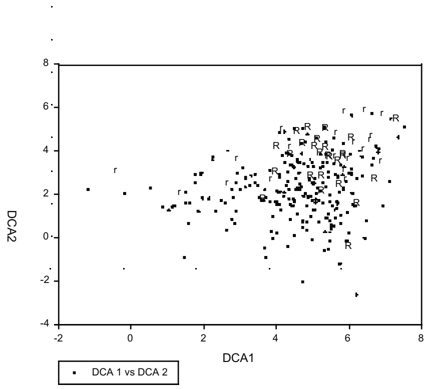


Figure 3.8 Species biplot of the first two DCA axes (r: rare species; R: rare river corridor species).

The rare river corridor plants are clearly grouped together with the dry river grasslands (Figure 3.8). The rare species of the dry river grasslands show a strong correlation with the third axis (Rare species-DCA3 z:5.74, p<0.001), indicating the isolated position of many river corridor plant relicts, situated far from the river and seldom flooded. Also, when considering environmental variables, the strongest covariance was observed between the number of rare species and the degree of isolation (Figure 3.9), which indicates that there is a group of highly isolated relicts harbouring a list of specific rare species. The well developed dry river grasslands are currently under great threat, and this is therefore particularly pertinent for the rare species of these communities. The problem of isolation is caused by an interruption of flood contact and habitat fragmentation due to changing land use conditions.

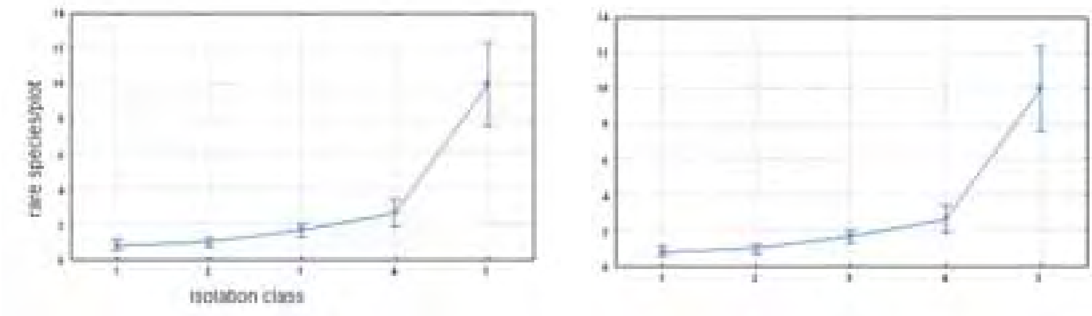


Figure 3.9 Numbers of rare species in the different isolation classes. This graph shows the significant covariance (ANOVA-result F=20,563, p<0,001) between isolation classes 1–5 and the number of rare species of the plots (182 plots with rare species).

Dry river grassland habitat conditions and modelling

Habitat conditions for dry river grasslands

The measured soil conditions for the 39 sampled dry river grasslands show that there is a distinct difference in these conditions depending on the community type. For example, high pH values (Figure 3.10) distinguish Sedo-Thymetum and Alysso-Sedion from Thero-Airion communities, and conductivity differentiates between the Sedo-thymetum and the Alysso-Sedion communities, as well as between the true Thero-Airion and the extended Thero-Airion+ group. For both pH and conductivity, the Medicagini-Avenetum grasslands take an intermediate position and have a broader range, which is obviously due to these grasslands occurring as a later succession stage in these pioneer communities.

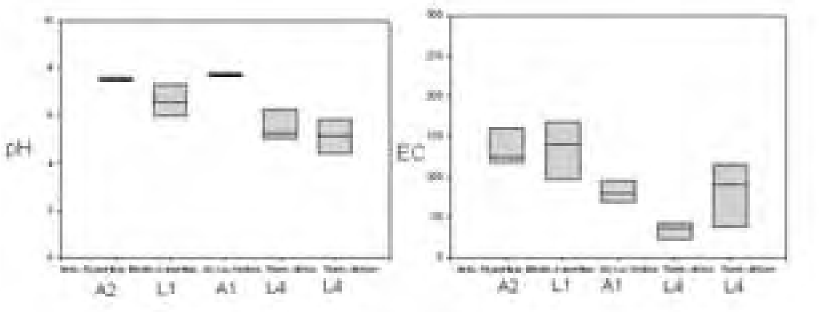
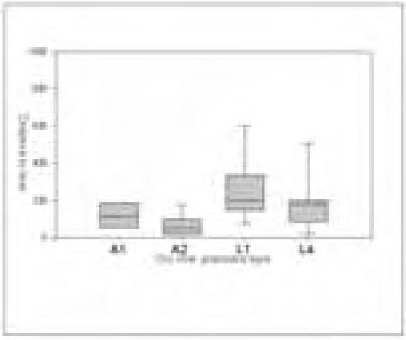
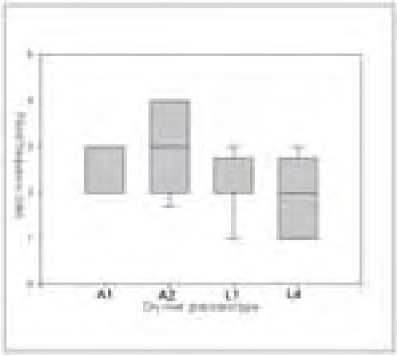


Figure 3.10 PH and conductivity of the dry river grassland communities sampled (Sedo-Thymetum, n= 7; Medicago-Avenetum, n=15; Alysso-Sedion, n=6; Thero-Airion, n=3; Thero-Air/Galio-Trifolietum, n=8). Boxes delineate 25th–75th percentiles; lines denote median values.

As correlations between these influential soil conditions (pH and conductivity) with flow regime and spatial patterns (flood frequency and distance to the river) were detected, sufficient predictive power was assumed present in the hydraulic modelling (flood frequency and flow velocity) to use these responses in a plant community prediction model. With the flow velocity determined as the strongest predictor variable for dry river grasslands in the CCA, and the flood frequency and distance to the river as important parameters with regard to soil conditions, we tried to parameterize these variables based on our dataset.

Using the observed relationships (Figures 3.11a-c), a dry river grassland model application was developed, in a stepwise integration of flood frequency, flow velocity and distance to the river as determining parameters. The combination of these three variables in a decision tree results in a higher predictive power than suggested in the charts.





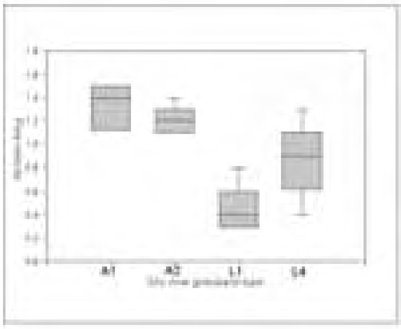


Figure 3.11 Box plots for the distance to the river (a), flood frequency class (b) and flow velocity (c) for the samples/plots of the dry river grassland types (boxes denote 25th–75th percentiles; bars denote the 95th percentile; median values are shown as lines).

The overbank gravel and sand depositions (A1, A2) are characteristic features of extreme flooding events and are restricted to less frequently flooded zones (flood frequency classes > 2; Figure 3.11b). So, they are retrieved from the model runs for extreme flood events (recurrence period 50 years) and determined in flow velocity classes (A1 Alysso-Sedion: >1.3 m/s; A2 Sedo-Thymetum: 0.9–1.3 m/s). The L1 and L4 types are divided in the modelling of the decennial flood (cfr. Fig. 9c), with flow velocity ranges of L1 Medicagini-Avenetum: 0.2–0.6; and L4 Thero-Airion: 0.6-1.1. Possible overlap is excluded with the third criterion (Figure 3.11a); the distance to the river discriminates between the A2 Sedo-Thymetum (< 80m) and the L4 Thero-Airion communities (> 80m).

The power of the model to predict the presence and potential development of dry river grasslands over the alluvial plain was verified by field testing, and this yielded good results (Figure 3.12). The modelling shows a potential of 158 ha of pioneer habitat creation, but only 19 ha was located in the field during vegetation mapping of the study area (Table 3.8). Yet, for the 56 patches predicted in the model, the inventory showed that nearly half of them (27) contained diagnostic species of these communities, mostly restricted to patch edges, road verges, etc. The intensive agricultural use of most of the valley explains this low correspondence.

Table 3.8 Modelling result and field survey validation of dry river grassland habitat.

Pioneer Patches	Potential modelled (ha)	Field mapping 2000 (ha)	Patches modelled (#)	Patches with diagnostic species
Overbank gravel deposition (A1)	44	11	23	10
Overbank sand deposition (A2, L4)	114	8	33	17

Figure 3.12 View of the model outcome for a part of the Common Meuse reach, with the predicted dry river grassland communities' potential distribution.

Discussion

Flood dynamics determine dry river grassland composition and diversity

Disturbance regime and flood contact are documented as playing a determining role in fluvial riparian vegetation (Bornette & Amoros 1996, Andersson et al. 2000). Direct relationships between community composition and flood regime and river contact were identified, as were indirect relationships through soil conditions. The development of communities is in the first place linked to aspects of flood regime, but further soil development and succession governs the community composition. The pioneer communities of Alysso-Sedion and Sedo-Thymetum are in a few years turned into Medicagini-Avenetum by the enrichment of the soil and the emergence of strong perennials. The Medicagini-Avenetum is the best known community for hosting river corridor plants. It can be described as a community in its optimum range because it covers such a broad range of soil and floodplain conditions of the dry river grasslands and it is only slowly replaced by other communities, a process that is sometimes quickened by grazing or fertilization practices.

Several authors have indicated flood intolerance (Jongman 1992, Grévilliot & Muller 2002, Eck et al. 2004, Vervuren et al. 2004, Leyer, 2005) and inadequate dispersal abilities (Hegland et al. 2001, Bischoff 2002, Vécrin et al. 2002, Donath et al. 2003, Stroh et al. 2005) as limiting factors for the restoration potential of dry river grasslands rich in river corridor species in the floodplains of larger Northwest European rivers. Our results do not conflict with these observations of distribution patterns and limitations with regard to flooding, but they do add another dimension in so far as flood events have also been proven as a crucial element in generating appropriate habitat conditions and dispersion potential.

The dry river grasslands have a strong preference for overbank depositions with a deep groundwater level, coarser texture and low organic matter content of the topsoil. Specific soil conditions govern the distribution pattern of dry river grassland communities; extremely high pH characterizes the Alysso-Sedion and Sedo-Thymetum pioneer communities of young river sediments, whereas the extremely low salt concentration of leached sediment soils characterizes the Thero-Airion communities in the valley. The correlation between pH, conductivity and flood contact indicates the important role of river flooding for these communities.

Rare species analysis and modelling result

The rare species analysis reveals important aspects of the threatened status of the dry river grassland communities and necessary measures for their conservation/restoration. There is a clear segregation of rare (river corridor) species of the dry river grasslands indicated by the ordination analysis. The significant covariance of diversity and the isolation of plots with richness in rare river corridor species is proof of their threatened status and a need for restoration. River corridor plants make up the majority of rare species and hence are good indicators of diversity and fragmentation at the river reach scale. The validation of the model revealed recruitment limitation as the main problem for the diagnostic species of these communities and their habitat. Therefore, a prerequisite when dealing with their isolation–fragmentation threats is habitat creation, linked to the periodic process of overbank deposition of sand–gravel sediments.

The flow variables proved the best predictors in the direct gradient analysis, useful in the elaboration of a model to predict the potential plant community distribution over the floodplain. The use of this kind of model application based on direct gradient analysis results works quite satisfactorily in this study, thanks largely to the restricted river reach scale level, and the basis of a validated two-dimensional river hydraulic model. Individual analysis of the influential variables adds strength to the relationships and predictions that were determined and allows a stepwise hierarchical modelling approach. Selection of predictors allows for a more accurate fit to the specific ecological niche of a community because the explained variance is much clearer in this way (Guisan et al. 1999).

Restoration potentials

Restoration projects in general aim at mitigating the effects of regulation works by rehabilitating geomorphologic processes to promote the recovery of degraded biota and the floodplain benefits from the river (Tockner & Schiemer 1997). However, the hydrological, geomorphologic and biological heterogeneity and variability of river floodplain systems, both temporally and spatially, sometimes complicate the restoration schemes (Amoros et al. 1987). Isolation in the river system has a spatial but, particularly, a temporal dimension, important for conclusions on conservation and restoration. The spatial and temporal habitat requirements of these communities can be seen in perspective of patch dynamics and shifting mosaic concepts (Petts & Bradley 1997, Barrat-Segretain & Amoros 1996). The shifting aspect of habitat is clearly linked to, and shows the intrinsic need for, river dynamics, as was also illustrated for these communities by Boedeltje et al. (2004) and Wolfert et al. (2002). If this study succeeds to recognize and parameterize this link between the physical and biological processes at temporal scales determined from flow variability, an effective restoration programme can evolve (Biggs et al. 2005).

The modelling approach – that integrated a set of combined rules of spatial and temporal prerequisites – provides a useful tool for identifying potential restoration sites and also an insight into the requirements for viable restoration of dry river grassland habitat and the characteristic river corridor plants. As the process of habitat creation does not occur annually, the spatial conclusions of the modelling also requires a temporal interpretation.

The potential for restoration – expressed in area for a characteristic unit of river length – could be defined as modelled area/recurrence period. For the studied Meuse reach, the 1/50 year peak discharge gives 158 ha of newly generated habitat of overbank sedimentation zones, with of course the restriction that existing habitat will be put back in succession as local stands will be over-deposited or eroded, allowing for seed and propagule dispersal but also provoking local extinctions. A 1/5 year peak generates 12ha. From these observations, a resulting restoration potential can be quantified as around 3 ha/year. This measure gives an idea of the necessary dynamics in conservation and management strategy.

Conservation strategy

Restrictions in land use or specific management strategies can allow the creation of new habitat but cannot stop gradual succession from pioneer to grassland communities, as these are governed by soil processes. Therefore, the pioneer communities only survive under the benefit of flooding events that generate the deposition of new sediments. Thus, the rehabilitation of fluvial processes is necessary to develop new habitat on the sites we determined through modelling. The rehabilitation of fluvial processes does not only mean that land use practices need to be changed, it also means that the river must transport enough coarse sediment. For this morphological criterion, sediment supply from eroding banks and larger gravel and sand bars in the river bed is necessary (Piégay et al. 2005). As these processes operate on a larger scale in space and time, a restoration approach at the reach scale is necessary to ensure the generation of new habitat for the future conservation of these vegetation types. In this way, from the present 19 ha, the realisation of 158 ha means a significant growth for an otherwise highly threatened habitat, and therefore a benefit for the river corridor species it contains.

Conclusion

Knowledge of the habitat creation process, together with the spatial and temporal requirements of the communities, allows for the design of effective restoration measures. The river corridor plants are a good flagship species group for the protection and restoration effort, as they cover a broad range of information on the characteristic habitats, and provide the best developed stands and richest vegetation. A modelling approach based on the analysis of community–environment relationships (CCA output) yielded management guidelines and demonstrated restoration potential for the dry river grasslands over the river reach.

III.3

Ground beetle habitat templets and riverbank integrity

Van Looy, K., Vanacker, S., Jochems, H., De Blust, G. & Dufrêne, M. 2005. Ground beetle habitat templets and riverbank integrity. *River Research & Applications vol. 21(10): 1-14.*

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Introduction

The appropriate application of ecological theories in the management of river systems, requires more knowledge of biological traits of riparian species (Barrat-Segretain, 1996). Where most river concepts focus on longitudinal patterns and gradients, the habitat templet theory is a useful approach for comparison and evaluation over and between river sections (Townsend et al., 1997). As organisms and communities in streambed landscapes respond to the type and spatial arrangement of habitat (Palmer et al., 2000, Eyre et al., 2001), the community responses of terrestrial riverine organisms are good predictors for river management impacts. River management has local effects on the spatial arrangement of habitats, but it can also generate downstream and upstream impacts on habitat integrity. Therefore, the sampling and analysis of biotic and abiotic parameters in river systems needs a hierarchic, scale-sensitive approach (Bovée, 1982, Frissell et al., 1986, Bauer, 1991, Gregory et al., 1991, Fawthrop, 1996, Petts & Bradley, 1997, Hansen et al. 1999).

The River Habitat Templet Theory (Townsend & Hildrew, 1994) offers a suitable approach to define indicators at river basin scale for habitat integrity and management. The use of habitat templets has benefits in defining responses and indicators in river systems with immediate relations to the physical conditions (Bornette et al., 1994, Townsend et al., 1997). Moreover ground beetles have been proposed as indicator group for river management (Sustek, 1994, Maiolini et al., 1998). Ground beetle assemblages have been recorded as responding to flood regimes (Bonn et al., 2002), riparian vegetation (Greenwood et al., 1995), riparian habitats (Boscaini et al., 1998, Plachter & Reich, 1998), riparian habitat heterogeneity and distribution (Eyre et al., 2001) and bank management (Gerken et al., 1991).

Ground beetles were chosen as bioindicators to assess the impact of flood protection strategies at Meuse riverbanks. The objective of this study was to identify responses of this species group to relevant parameters for the river management, to be integrated in an evaluation method for flood protection and river restoration. Carabid beetle assemblages were determined along the river's longitudinal gradient, and indicators for habitat integrity were derived from clustering and nestedness analysis of species assemblages. Further responses to river management related variables were identified with multivariate analysis. The important species traits for the templets were linked to habitat use and selection. Responses to specific river conditions of these templets are useful in the evaluation of river management and flood protection measures in particular.

Methods

Study area

The river Meuse is a rain-fed river, originating at an altitude of 409 m above sea level at the Plateau of Langres in the North of France and discharging into the North Sea some 900 km further downstream. The catchment's area is c. 33,000 km?, situated in France (9,000 km^2), Belgium (13,500 km^2), Germany (4,000 km?), Luxembourg (600 km^2) and the Netherlands (6,000 km^2). As the research focused large river's bank habitats, some 400km of the river's middle course were investigated. A detailed survey was executed in the unregulated middle course section (50 km) of the Common Meuse between the towns of Maastricht and Maaseik on the Flemish-Dutch border. The Common Meuse is a gravel river with a strong longitudinal gradient (0.45m/km). The discharge of this rain-fed river shows great fluctuations. Discharge levels for the Common Meuse range from 10 m^3/s during dry periods to 3,000 m^2/s in periods of heavy rainfall.

Studied species

Ground beetles have a wide range of ecological traits, related to habitat conditions of food supply, substrate and vegetation cover. Species traits of wing development, dorsal flattening, reproduction rhythms and phenology mean that ground beetles are very selective in terms of habitat affinities (Den Boer et al., 1979, Desender et al., 1994). Their potential use as bioindicators in surveys of riverbank communities along the Meuse and its main tributaries has been discussed previously (Baufays, 1994, Dufrêne & Legendre, 1997, Richir, 2000). Ground beetles are commonly referred to as a good indicator group as they exhibit habitat selection, varying dispersal capacity and colonising strategies (Stork, 1990). Moreover, the family is taxonomically well known and easily sampled. The combination of these abilities, together with the large number of river species, allows the distinction of indicator groups for environmental characteristics, habitat configuration and integrity in river systems (Zulka, 1994), and even for larger rivers in a global context (Boscaini et al., 1998).

Sampling

Data on the riparian carabid fauna and vegetation were collected during 3 consecutive years 1998-2000 along the river Meuse. The sampling at a catchment scale was executed in 2000 using 14 stations spread along the middle to lower course of the river Meuse (Figure 3.13). The reach scale sampling of the Common Meuse was carried out for two consecutive years 1998 and 1999 on 17 gravel bank stations sampled of the 50 km Common Meuse river reach. Each station consisted of two plots; one higher on the riverbank and one close to the waterline, giving 34 plots in total. Ground beetles were sampled using pitfall traps (filled with 5% formaldehyde preservative), three traps in a row at 1m intervals forming a plot. Samples from the 3 traps were pooled and species identified in the laboratory. The traps were sampled in two-weekly intervals for the period of May to October in both years. Although not without problems, pitfall sampling has been used extensively to compare species assemblages in larger geographical areas under river bank conditions (Dufrêne 1992, Spence & Niemelä, 1994, Desender & Maelfait, 1999, Eyre & Luff, 2002). However, abundance and especially size-abundance relationships require careful interpretation (Arneberg & Andersen, 2003).

Together with the biotic sampling, data on river bank and habitat characteristics were collected and stored as catchment and reach scale river variables (Table 3.9). This set of independent variables was retained from a broad range of variables, selected from relevant literature (Armitage et al., 2001, Growns & Growns, 2001, Bonn et al., 2002, Olden & Poff 2003). For the different gauging stations (Stenay, Lorraine Meuse / Ampsin-Neuville, Ardennes Meuse / Borgharen, Common Meuse / Venlo, Sand Meuse) data of 10 year average daily discharges were used to derive the hydrological indices at the catchment scale. For the Common Meuse, hourly flow data of the last 10 years were analysed. The selected hydrological indices are widely used in the description of flow modifications, especially in flow regulation assessment (Growns & Growns, 2001). The following definitions were used: baseflow index (BFI) = (lowest daily discharge/mean daily discharge) x 100, coefficient of variation (CV) = (standard deviation of monthly discharge/mean monthly discharge) x 100, peak frequency (PF) = number of discrete flood events, i.c. the peak fluxes (when discharges exceed the level of the riverbank dynamic habitats) during the summer period (may to October, the active period for the carabid fauna), peak velocity (PV) = the peak flux over hourly discharges, derived from the summer peak events over the longyear flow data, and the rising speed (RS) = the velocity of the water level rise, defined as the difference in water level (m) between 200 m^3/s and 10 m^3/s discharge as a measure for the hydrodynamics on the riverbank habitat.

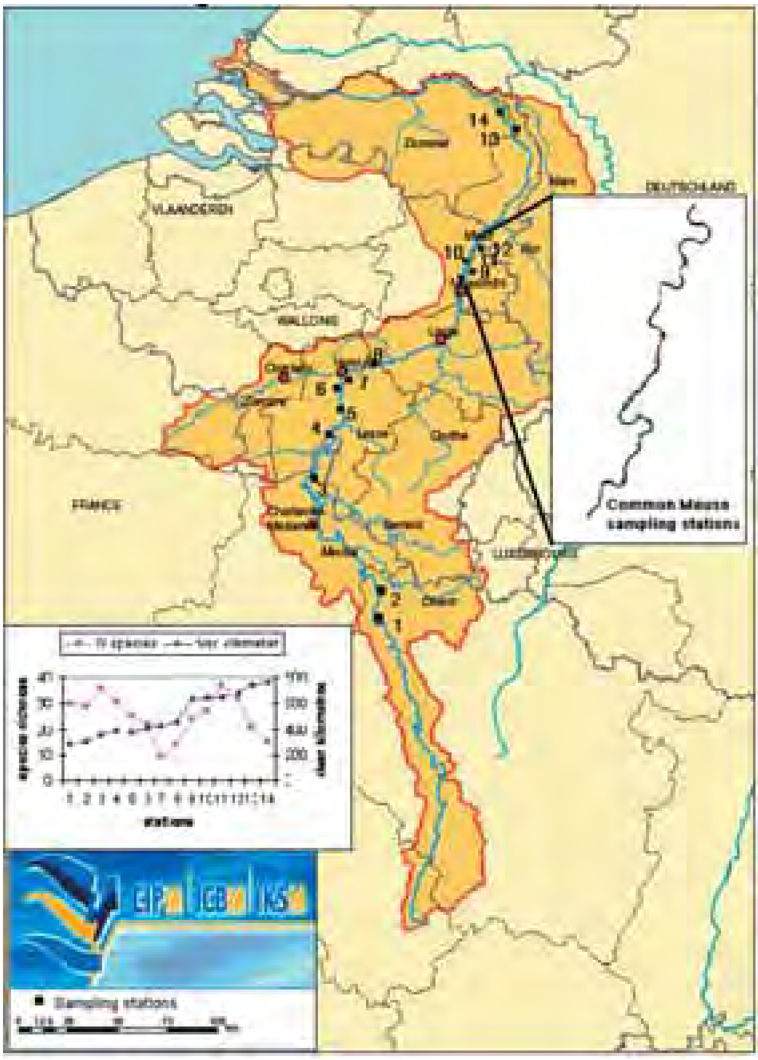

Figure 3.13 The Meuse river basin map of the International Meuse Commission with the Carabid sampling stations (with inset for reach level sampling of the Common Meuse stretch) and their richness in habitat templet indicator species in the inserted diagram.

Table 3.9 River variables of channel morphology, hydrology and bank characteristics as surveyed for the catchment and reach level sampling.

VARIABLE	DESCRIPTION	MEASUREMENT
River kilometre	Distance from river source (km)	Catchment/reach
Width/Depth–ratio	Dividing river width by mean river depth	Catchment/reach
Baseflow index	Dividing lowest flow by mean flow	Catchment
Coefficient of Variation	Dividing discharge variation by mean discharge	Catchment
Peak frequency (of summer peaks)	Number of relevant summer peaks in summer season	Catchment
Peak velocity	Hourly or daily maximum flow increment	Reach
Rising Speed	Velocity of water level rise	Reach
Habitat heterogeneity	Number of habitat types per station (within 20m around plots)	Catchment/reach
Texture	D50-value of substrate (mm)	Catchment/reach
Vegetation cover	Percentage soil covered by plants (%)	Catchment/reach

Some further variables, relevant in riverbank habitat description, were included: river kilometre, width/depth-ratio, habitat heterogeneity (# bank habitat types per station), texture of substrate and vegetation cover. Vegetation sampling occurred in a mapping of vegetation types in a range of 20 m around the plot and a 1 m? relevee at the plot site. These data were used for the definition of habitat heterogeneity of the stations, while the coverage of the relevees was used for the vegetation cover parameter.

Analyses

The habitat templets were derived from a clustering and ordination of species assemblages from the catchment level sampling set (Figure 3.14). The plot-species matrix was selected on species (>3 individuals), plots (>80 individuals per plot) and plots/species (>2 plots/species). 16 plots and 77 species were retained for the analysis at the catchment's scale, 29 plots and 84 species for the reach scale.

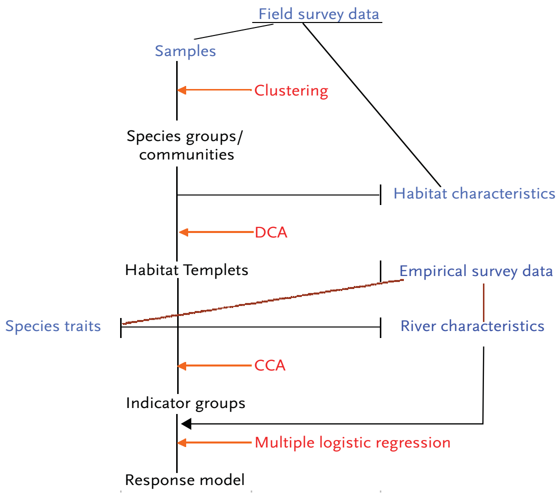

Figure 3.14 Flowchart of habitat templet approach.

For the classification of faunistic site sampling data, a non-hierarchical clustering method is the most appropriate (Dufrêne & Legendre, 1997). The k-means program (Legendre & Vaudor, 1991) is a least partitioning method that divides a collection of data into 'k' groups. The algorithm computes clusters and assigns each species to the nearest cluster at each level of k, in such a way as to maximize the between-cluster differences.

Before entering the clustering program, a principal coordinate analysis (PCoA) was run from the similarity matrix, using the Steinhaus coefficient (Legendre & Legendre, 1983), calculated on natural log-transformed data. The k-means method was applied to the plot coordinates on the first 12 PCoA axes of the Steinhaus similarity matrix, allowing the filtering of the ordination axes and the identification of a hierarchical structure in the data if present (Dufrêne & Legendre, 1997).

Together with the clustering, the identification of indicator species with the IndVal-method (Dufrêne & Legendre, 1997) computed indicator values at each level of 'k'. The INDVAL-index is maximal (100%) when all individuals of a species are observed in all sites of that site-group. The IndVal indicator value is not only a reliable measure in the proposed clustering method, but is a absolute measure, making comparisons across taxa, functional groups and communities robust to differences in abundance (McGeoch & Chown, 1998). Indicator species with high fidelity and specificity were selected for each habitat templet.

An ordination by DCA was computed in the CANOCO program (ter Braak, 1988). Based upon the length of the DCA gradient-value, a Canonical correspondence analysis (CCA) was performed with the environmental variables included. A first set of variables, relevant for ecological effect assessment of flow regulation (Growns & Growns, 2001), was determined at the catchment's scale-level (Table 3.9). Further analysis of the hydrological parameters was done at the reach scale to detect responses to hydrological regime (in-between years and reach plots) and management parameters.

With the detected predictor variables for the riverbank carabid faunal composition, a covariance analysis was run for the templet indicator species. In the STATISTICA program, the datasets of the catchment and reach level sampling were analysed with non-parametric tests for 2 independent samples (Mann-Whitney and Wilcoxon). Covariance between the habitat templet indicator species and species richness of the plots was analysed (with Mann-Whitney test). Before entering this covariance analysis, a nestedness analysis was run, to detect matrix temperature and nested subsets with the Nestedness Temperature Calculator Program (Atmar & Patterson, 1995). Nestedness is a way to estimate the degree of hierarchy in species assemblages, which allows the distinction of indicators for species richness in hierarchic sets (Atmar & Patterson, 1993, Worthen, 1996, Gustafsson, 2000, Honnay et al., 1999). Strong covariance detects those templet indicator species that are good indicators for the biotic integrity of the riverbank. To conclude the correlation analysis, Mann-Whitney covariance testing was done on the datasets for the river management variables at the two dataset levels. The dependent (grouping) variables were the presence-absences of the indicator species, the species richness and the habitat heterogeneity respectively.
As the final step a multiple logistic regression was executed on the detected indicator species for the river management variables width/depth ratio, peak frequency and peak velocity. From this logistic regression a response and optimum range of the variables for the biological integrity, was assessed.

Results

The catchment sampling yielded 4,892 ground beetles extracted from the pitfalls and determined to species level (Table 3.10). Over 16,000 carabid beetles were sampled and determined from the 1998-1999 Common Meuse reach level sampling. The k-means clustering of the samples similarity coordinates gave the best fit for eight species groups (Figure 3.15). At each level below level 8 the species with the highest INDVAL-value are listed. At level 8 the cluster groups are shown with all species with INDVAL-values > 25 per group included. The clustering separates the sites closest to the waterline from the more elevated sites. The further differentiation accords to the present substrate and vegetation cover.

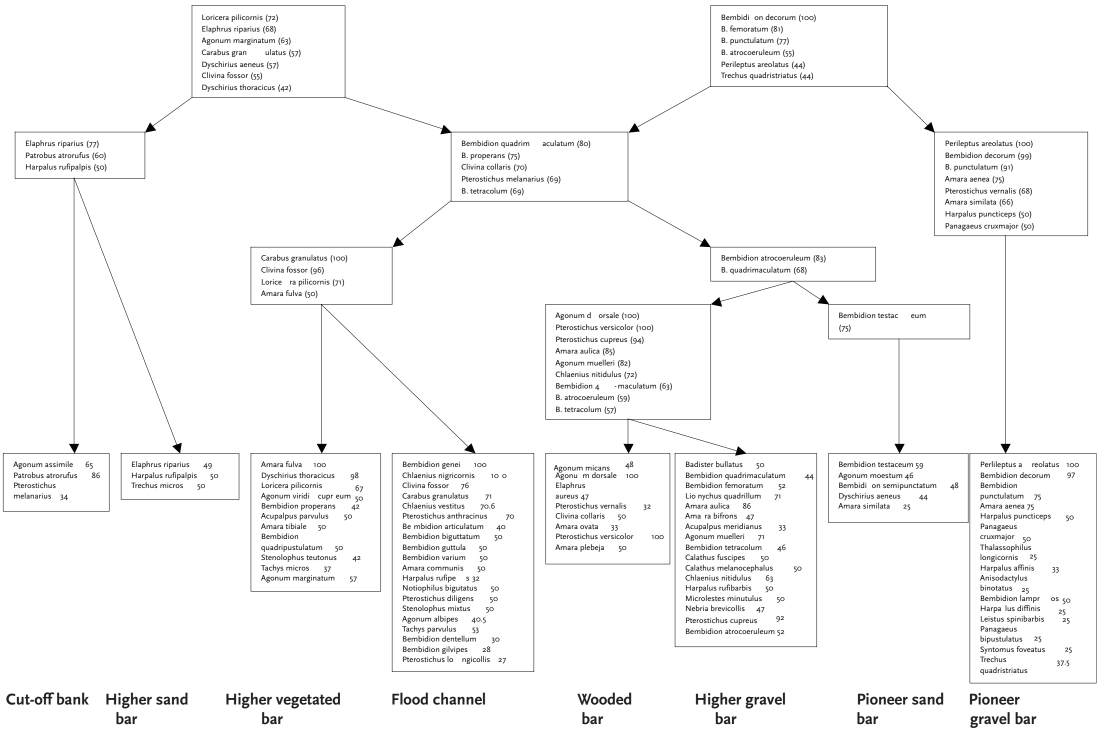

Figure 3.15 Site clusters with templet indicator species groups (INDVAL values > 25) obtained with the k-means method. For the hierarchic divisions the species associated INDVAL indicator values are given in parentheses

The Detrended Correspondence Analysis (Figure 3.16) shows a strong influence of the river dynamics along axis 2, from the pioneer bars to the flood channel plots. The axis 1 division is related to the naturalness/modification of the riverbank, with the riverbank related species situated to the left, and the eurytope species to the right. The influence and inflow of species from adjacent fields dominates more to the right.

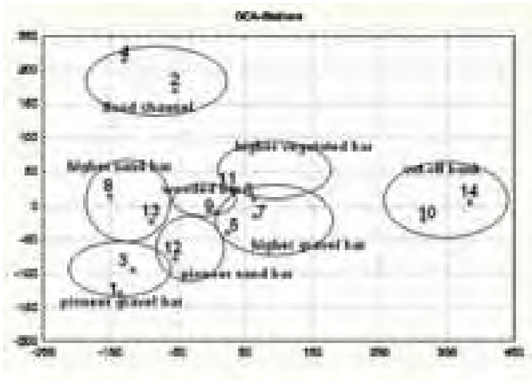

Figure 3.16 DCA-plot of the 16 sampling plots with the confidence ellipses for the 8 habitat templets.

In Figure 3.17 the triplot for the Canonical Correspondence Analysis (CCA) is given for the dataset of the catchment level sampling. The first axis in the CCA explained 36.8% of the total variance and coïncided for 91% with the variable width/depth-ratio, to a lesser extent with peak frequency. The second axis added 37.8% to the explanatory value, and was correlated for 80% with soil texture.
A high correlation with width/depth-ratio and peak frequency was observed for the 'pioneer gravel bar' indicator species. High habitat selectivity of this group was already shown in the INDVAL values of the indicator species (Bembidion punctulatum INDVAL 75.27, Bembidion decorum 96.82, Perileptus areolatus 100 and Amara aenea 75). This templet shows a negative correlation with vegetation-cover. The templets 'flood channel', 'pioneer sand bar' and 'higher vegetated bar' were correlated with sandy texture and high vegetation-cover. Species associated with the river kilometre variable were only few, restricted to the sampling of downstream (Patrobus atrorufus, Agonum assimile) or upstream (Bembidion dentellum and Harpalus puncticeps) stations.

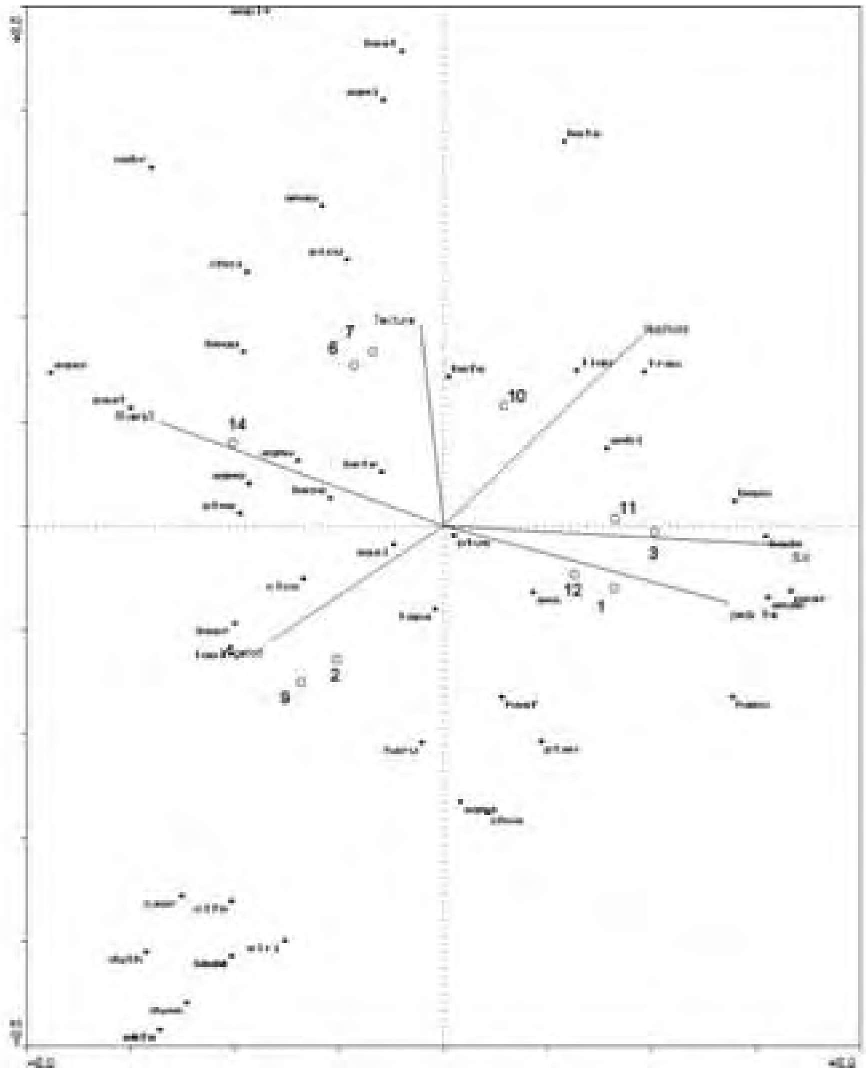

Figure 3.17 Canonical correspondence analysis (CCA) triplot of carabid species, sampling stations and environmental variables along 600km of the river Meuse (sampling 2000).

Figure 3.18 Canonical correspondence analysis (CCA) biplot of carabid species (1998-1999 sampling) and 4 environmental variables

Figure 3.18 shows the result of the correspondence analysis at the reach level. The strongest correlations were detected with the peak velocity (with the first axis 82%), and to a lesser extent rising speed (for the fourth axis 81%). Width/depth ratio showed a high correlation with the higher elevation habitat templets (higher vegetated bar and higher gravel bar). Highest habitat heterogeneity was observed for the 'flood channel' and 'wooded bar' templets, as they are only present in the most natural stations.

Response analysis

The INDVAL determined habitat templet indicator species were entered in the analysis for riverbank integrity indicator species. Indicators for the habitat

integrity of the riverbank as a whole were detected in the nestedness analysis. A significant covariance (Wilcoxon p value 0.00019) was first detected between species richness and habitat heterogeneity. The assembly of ground beetles at the catchment as well as at the reach scale had a significantly nested structure. For the reach sampling, the data set temperature of 10.64° deviated significantly ($p< 0.001$) from the simulated set temperature of 39.5°. For the catchment's data, the matrix temperature was 32.27°, indicating the wider spreading of the species data, but still significantly deviating ($p<0.001$) from the Monte Carlo simulation run in the Temperature Calculation Program.
So, Mann-Whitney covariance testing allowed detection of indicators for the biotic integrity of the Meuse riverbanks. Overall Meuse riverbank bioindicators (Bembidion tetracolum, Chlaenius nitidulus, Pterostichus vernalis, Amara similata and Harpalus affinis) were detected in the correlation with species richness (Table 3.10).

Indicator species and river management variables

For the species richness, significant correlations were detected with peak velocity (explained beta-variation: 0.36) at the reach level, and with peak frequency at the catchment level (explained beta-variation: 0.47). The W/d ratio covariance was significant for 21 habitat templet indicator species. The species with the strongest significant covariance and CCA correspondence values preferred the broader stretches with W/d ratios above 25.
The indicator species with the strongest covariance for the peak frequency are Perileptus areolatus (chi^2: 11.4, $p=0.0007$) and Amara aenea (chi^2: 7.9, $p=0.004$). For the indicator species, optimum peak frequency lies in the observed maximum of 9 summer peaks.
The indicator species for the peak velocity (Harpalus affinis $chi^2= 25.9$, $p<0.0000004$ and Bembidion decorum $chi^2= 22.1$, $p<0.0000026$), showed an optimum below 30 in the logistic regression.

Discussion

The use of single species or taxonomic groups as indicators for the integrity or quality of ecosystems has been criticized (Landres et al., 1988; Niemi et al., 1997;

Prendergast et al., 1993) because the effectiveness of the concept has often been assumed, but only rarely tested (Andersen, 1999, McGeoch & Chown, 1998). Furthermore, the selection of bioindicators for river health assessment needs a scale-sensitive survey and analysis of distribution and selection of habitat (Fairweather, 1999, Karr 1999, Norris & Thoms 1999, Hansen et al., 1999, Pedroli et al., 2002). The broad range of species traits and habitat adaptations makes ground beetles a good candidate indicator group for habitat integrity and river health assessment in general. The habitat selectivity is reflected in the species traits as the smallest, flattest, flying species are best adapted to the most dynamic riverbank habitats (Desender, 1989, Eyre & Luff, 2001). Larger, slower species of the genus Carabus or Pterostichus are restricted to the higher, less dynamic zones. The clear segregation of habitats in the riverbank, caused by sharp boundaries of substrate and vegetation cover, contributes to the high INDVAL-values for the habitat indicator species.
The relevance of carabid beetles as bioindicators for hydromorphological processes and riverbank habitat integrity was already tested in local as well as global river management context, using the same sampling method (Boscaini et al. 1998, Maiolini et al., 1998, Kleinwächter et al. 2003). The identified habitat templets include a large number of riverbank species with high INDVAL-values. These are valuable bioindicators for the riparian habitats, as the INDVAL method selects species more or less unique to the habitat (high specificity) as well as widespread within it (high fidelity). So these indicator species have not only high information content, but also a high probability of being sampled during monitoring and assessment. This habitat specificity does not imply that the identified indicator species in our riverbank survey are restricted to riparian habitats. Several generalists of open and disturbed ground were attributed to specific riparian habitats. The same observation can be made for the use of the riparian zone in other organism groups.
The presence of many habitat specialists in this organism group for the riparian zone, contrasts with Meuse macroinvertebrate surveys. In a macroinvertebrate sampling of the littoral zone of the Common Meuse, only 1% of the sampled individuals was habitat specialist (Smit & Gardeniers, 1986). This distortion was attributed to the strong anthropogenic disturbances in the habitat conditions related to flow regime and mainly water quality. As many carabid species inhabit the summer bed in low flow conditions, anthropogenic changes

to the river system are also reflected in changes in species composition. Indeed in our dataset, the assemblages of highly modified banks contain the smallest number of indicator species, while more undisturbed stations have more templet indicator species. Nevertheless, with more than 90% of the individuals belonging to riverbank habitat templet indicator species, the abundance of riparian habitat specialists in our sampling set was spectacular. So, where dramatic changes in aquatic organism groups were caused by anthropogenic disturbances, terrestrial riverbed habitats still preserved characteristic communities allowing river health assessment for the hydromorphic aspects.

The significant correlation between the habitat heterogeneity and the species diversity is important for the habitat integrity assessment. The heterogeneity in riverbank habitats yields more potential ecological niches to be filled at the same location (Sadler et al., 2004). The richness of templet indicator species over the pitfall sampling stations along the Meuse shows the lower integrity in the Ardennes Meuse and Sand Meuse stations (inset in Figure 1). The heavily regulated Belgian and Dutch Meuse reaches show a drastic decline of stream integrity, with a strong recovery in the un-navigable Common Meuse reach. Stream canalisation efforts for navigation in the Ardennes and Sand Meuse, with embankments and groins, reduced the available riparian habitats for terrestrial as well as aquatic macroinvertebrate communities dramatically (Usseglio-Polatera & Beisel, 2002).

No clear shift in communities along the river was observed for the ground beetles in contrast with the longitudinal changes in macroinvertebrate assemblages and alleged problems for the coordinated assessment of the biological integrity for the whole river Meuse (Usseglio-Polatera & Beisel, 2002). So, with the defined habitat templets, we can work out an unbiased catchment's scale 'river health' bioassessment.

The need for quantification of physical and biological responses remains a main issue for the evaluation of river management and flood protection measures (Van Kalken & Havno, 1992, Large & Petts, 1996, Pedroli et al., 2002). To adequately describe the main aspects of the flow regime and relevant biological consequences, the use of different hydrological indices is required (Olden & Poff , 2003). Also the need for multi-scale approaches in river ecology and restoration is stressed (Wiens, 1989, Hansen et al., 1999, Rabeni & Sowa, 2000). Gathering the necessary data requires extensive work and the same counts for the data screening and detection of significant correlations and responses.

The determining variables in the clustering and ordination identified here are generally applicable flood regime and riverbank management parameters, and can serve as predictor variables over reaches and even in-between rivers. Surveys of major German rivers (Bonn et al., 2002) and exposed riverine sediments in Scotland and England (Eyre & Luff, 2002, Sadler et al., 2004) showed separations based on differences in flooding regime and habitat conditions similar to our conclusions. The important key predictor variables were width-depth ratio and peak frequency/velocity and both are widely used variables in the description of river dynamic character and river management. The responses of the ground beetle community to river management practices can be successfully evaluated based on our results. The main explanatory variables of bed profile and habitat heterogeneity indicate the responses to management practices of riverbed widening and bank lowering in a positive sense, and encroachment and embankments in a more negative way. Although the strongest determining parameters are associated with the spatial facets of habitat availability, the indices of flow regime added a complementary set of explanatory variables for the ground beetle communities. Hence, the hydrological management on the river basin level is a trigger factor for the riparian biota and regulation activities, weir management and retention strategies have impact on the biological integrity of riverbanks throughout the whole river basin.

Conclusion

Research and evaluation tools in flood protection and river restoration projects focus mainly on hydrological relationships, only recently the geomorphic aspect has gained attention. The presented habitat templet approach envisages the hydromorphological impact on the riverbank, based on habitat and species group traits. Apart from water level effect prediction, a set of parameters describing peak characteristics and morphodynamics should at least be estimated in evaluation methods. Responses to a set of hydrological and morphological parameters were identified that allow riverbank habitat integrity assessment. From the presented analysis, an evaluation tool was elaborated (Geilen et al., 2001) that is not solely focused on the intrinsic quality of riverbank habitat, but at the same time allows qualitative assessment of impacts, on the spot as well as downstream and upstream by responses to hydromorphological parameters.

Abstract

The habitat templet approach was used in a scale-sensitive bioindicator assessment for the ecological integrity of riverbanks and for specific responses to river management. Ground beetle habitat templets were derived from a catchment scale sampling, integrating the overall variety of bank types. This coarse-filter analysis was integrated in the reach scale fine-filtering approaches of community responses to habitat integrity and river management impacts. Higher species diversity was associated with the higher heterogeneity in bank habitats of the unnavigable river reaches. The abundant presence of habitat specialists in the riverbank zone, allows a habitat integrity assessment based on the habitat templet indicator species. Significant responses were detected for channel morphology in the width/depth ratio and for hydrological regime in peak frequency and peak velocity, enabling the development of evaluation methods for the impact assessment of river management and flood protection strategies.

Acknowledgements

The assessment of riverbank habitat integrity was embedded in the European Commission funded international research program on flood protection measures for the Rhine and the Meuse, the IRMA-SPONGE Intermeuse project (Geilen et al. 2001). The selection and inter-correlation of the predictive variables with their critical ranges resulted in an evaluation method for flood protection strategies. The information of the gauging stations was provided for the analysis in this ecological assessment, with the permission of the management authorities, the Directorate Limburg for the Netherlands, the Administration of waterways and sea, DIHO for Flanders and the Direction Regional Lorraine for France.

Table 3.10 Mann-Whittney test for covariance with species richness, habitat heterogeneity and width/depth ratio of the plots, ** significant p<0,05, * 0,05<p<0,1.

species		Agonum albipes	Agonum assimile	Agonum dorsale	Agonum margina	Agonum micans	Agonum moestu	Agonum muelleri	Amara aenea	Amara aulica	Amara bifrons
total:		69	68	30	269	27	7	105	5	5	4
stations:		6	4	2	12	3	4	5	3	4	2
	species richness	0,81	0,63	0,43	0,39	0,9	0,67	0,62	0,34	0,3	0,77
habitat hetero-geneity	0,000197**	0,28	0,75	0,57	0,16	0,44	0,49	0,92	0,09*	0,7	0,57
W/d ratio	0,57	0,63	0,45	0,39	0,42	0,95	0,079	0,45	0,009**	0,8	0,26

species	chlaenius nigricornis	Chlaen nitidulus	Chlaen vestitus	Clivina collaris	Clivina fossor	Dyschir aeneus	Dyschir thoracic	Elaphr riparius	Harpa affinis	Harpa punctic	Harpa rufipes
Total:	16	7	10	26	33	7	57	21	96	4	38
stations:	2	4	5	7	6	4	4	6	7	2	6
species richness	1	0,06*	0,65	0,38	0,17	0,52	0,79	0,63	0,05*	0,36	0,13
habitat hetero-geneity	1	0,39	0,84	0,41	0,96	0,12	0,42	0,26	0,18	0,57	0,7
W/d ratio	0,16	0,2	0,55	0,12	0,39	0,26	0,75	0,77	0,36	0,09*	0,92

Amara fulva	Amara similata	Bembid articulat	Bembid atrocoer	Bembid decoru	Bembid femorat	Bembid propera	Bembid punctul	Bembid quadrim	Bembid semipu	Bembidtetr acolum	Bemb tes- tac	Carabu granulatu
5	9	5	19	1818	313	91	866	47	7	244	46	15
2	8	2	5	9	14	11	9	10	5	14	3	5
0,09*	0,05*	1	0,96	0,42	0,6	0,71	0,69	0,65	0,96	0,06*	0,85	0,32
1	0,006**	1	0,34	0,56	0,26	0,89	0,12	0,72	0,37	0,59	0,28	0,88
0,6	0,029**	0,16	0,21	0,17	0,49	0,44	0,03**	0,18	0,16	0,95	0,63	0,19

Lionych quadrill	Loricer pil- icorni	Nebria brevicol	Patrob atrorufu	Perilep areolatu	Pterost anthrac	Pterost cupreus	Ptero melanar	Pterost vernalis	Pterost versicol	Tachys parvulus	Stenol teuton	Trechu quadristr
5	55	36	41	32	13	69	116	18	19	9	6	34
3	14	4	8	4	5	9	13	11	2	5	2	5
0,4	0,67	0,36	0,75	0,36	0,43	0,72	0,11	0,04**	0,43	0,49	0,77	0,76
0,81	0,04	0,26	0,53	0,08*	0,16	0,89	0,88	0,03**	0,57	0,84	1	0,76
0,05*	0,009	0,79	0,37	0,002**	0,23	0,72	0,21	0,61	0,39	0,55	0,16	0,12

IV Tresholds and boundary conditions

In this part, responses and boundary conditions are investigated, as we aim at revealing/understanding relationships between the physical and biotic system, and quality assessment and measures for this relation, we need knowledge on thresholds and response functions for the defined key elements. These we identified as: riparian forest and ground beetles, and river corridor plants of dry river grasslands in the floodplain. The response analysis goes beyond the level of communities and biodiversity in general, as it focuses on specific and significant thresholds and boundary conditions detectable for critical species.

The surveyed scale levels for this chapter are the river reach, and more specifically the Common Meuse reach, in combination with the local site level.

Within the processes the scope is for the flow regime characteristics at low flows as well as peak discharges, for the management practices in the riparian zone as well as in the floodplain and for the biotic processes at dispersal and recruitment, fragmentation and isolation at the different scale levels.

Central questions are for the potentials for recovery of threatened communities and species and the organising role played by fluvial processes in the restoration potentials.

Themes and groups emphasized upon are riparian forests for the larger structures at river reach scale, ground beetles for the small-scale responses to critical processes at reach scale, yet at smaller dimensions of low flows. For the critical factors in the floodplain the river corridor plants are key elements.

IV.1

Predicting patterns of riparian forest restoration.

Van Looy K., Severyns J., Jochems H. and De Smedt F. 2005. Predicting patterns of riparian forest restoration. Large Rivers Vol. 15, No. 1-4, Arch. Hydrolbiol. Suppl. 155/1-4, p. 373-390.

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Abstract

The river's morphodynamic processes are an intrinsic aspect of riparian forest development. Sedimentation and bar formation are prerequisites for the stages of germination and growth of riparian forests. Furthermore, the mechanical disturbance of plants by erosion and abrasion, define the boundary conditions of establishing riparian forests. A field survey and hydraulic modelling of a 17 km river stretch revealed the patterns and processes of forest development in relation to hydromorphological and biological characteristics. These patterns were introduced in a modelling and prediction of riparian forest development within spatio-temporal sequences. The determined physical and biological components in forest restoration allow us to give guidelines for restoration strategies and plans at the different scale levels.

Introduction

The restoration of riparian forests is one of the main objectives of river rehabilitation projects around the world. In unmodified river systems, riparian vegetation exhibits a zonation from the river channel to the uplands along an elevation gradient (Lyon & Sagers 1998, Pabst & Spies 1998, Pautou & Wuillot 1989, Siebel & Bouwma 1998). Where regulation and engineering works disturb the regular flooding and vegetation patterns of the fluvial system (Bravard et al. 1986, Carbiener & Schnitzler 1990, Carter Johnson 1997, Shafroth et al. 2002), rehabilitation projects focus on enabling river dynamic processes that maintain the floodplain habitat heterogeneity. For the regeneration of riparian forests, flooding events are documented as an essential feature (Schnitzler 1997, Baumgärtel & Zehm 1999, Hughes et al. 2001, Bovee & Scott 2002). From the river manager's point of view flow resistance in space and time is a crucial aspect for riparian forest restoration. Therefore the prediction of forest development with emphasis on age structure and location is an important element in restoration programmes. Softwood forests contribute, by their rapid growth and strong flow resistance, to raising bar and island levels by retaining sand and gravel. The bars and islands in formation grow regularly in width and height with continuing accumulation of trapped sediment and an encroachment of willow thickets.
Hydraulic modelling is an extremely expanding science with strongly reliable

measurements of flow resistance and erosion-sedimentation processes under specific riverbed conditions. The integration of vegetation dynamics in these modelling approaches is still unexplored. Aim of the research was to identify spatial and temporal patterns of riparian forest development, applicable in modelling of forest development. In this paper the role of morphodynamic processes in the development of the Salici-Populetum river forests is quantified in critical ranges/thresholds of hydraulic parameters, allowing predictions of forest development after restoration.

Material and method

Studied river reach

The Meuse is a rain-fed river, originating at an altitude of 409 m above sea level at the Plateau of Langres in the North of France and discharging into the North Sea some 900 km further downstream. The mostly rocky underground of the upstream part of the catchment, explains the rain-fed character with high discharge fluctuations for the Common Meuse. The Common Meuse is the 45 km Flemish-Dutch border section of the Meuse between Maastricht and Maaseik. It is a unregulated gravel bed river with a high slope (0.45 m/km). Discharge levels for the Common Meuse range from 10 m^3/s during dry periods to 3.000 m^3/s in periods of heavy rainfall in the catchment. The studied reach is a 17 km stretch of the Common Meuse between Smeermaas and Maasband.
For the Common Meuse, a large-scale restoration project is defined aiming to rehabilitate the river's gravel bed and to restore flood contact with the alluvial plain (Van Leussen et al. 2000, Pedroli et al. 2002, Van Deursen et al. 2001). River regulation activities and fragmentation of valley ecosystems were identified as strongly affecting diversity and composition of the Common Meuse river forests (Van Looy et al. 2003) and immediately threatening gene flow and genetic diversity resources (Bunn & Hughes 1997, Imbert & Lefèvre 2003). Despite the absence of flow regulation and shipping on the Common Meuse river stretch, bank reinforcement and former gravel mining in the river bed resulted in a strong decline of the morphological activity and of the presence of bars and islands (Micha & Borlée 1989). However, since 10 years the bank management changed with the adoption of the

restoration programme and the growing awareness of new approaches in flood management. For this research bank erosion and forest settlement were allowed within safety limits. With the major flood events of 1993 and 1995, the morphological activity of the river reach showed a strong revitalisation in the elevation and reforestation of bars and islands.

Field and map survey

The development of pristine riparian forests of Salix and Populus species on the deforested Common Meuse stretch were surveyed for its morphological and biotic characteristics. The recruitment and age of willow and poplar trees was recorded for four consecutive years (1998-2001) along the study reach. All banks and bars were visited in September and all woody species were identified and measured. The age of the trees on bars, islands and the riverbank zone was determined on morphological aspects (year sprouts for young trees and ring detection with Pressler bore for older trees). Individual trees and seedlings were mapped. For developing forests, the coverage and age-classes of the different species were recorded.
Along with the woody species, erosion-sedimentation rates of bars and islands were described for the 4 years. The highest point of the bar was marked (at a tree) and the difference over the years was described as sedimentation (for elevation) or erosion (for lowering).
From bed profile measurements (1930-1987-1997) and aerial photographs (1990, 1995, 1996 and 2000) the age of bars and islands was investigated, together with the delineation of earlier bank line position (from the profiles), in order to describe the bank retreat process.

Spatio-temporal sequences

The regeneration of riparian forest can be determined in spatio-temporal sequences; this proved a valuable approach in integrated process and pattern analyses (Klein et al. 1995, Bartha et al. 1997, Ward & Stanford 1995, Pautou et al. 1997, Chiarello & Barrat-Segretain. 1997, Verheyen & Hermy 2001).
Temporal and spatial sequences were derived from the field and map survey.

From a GIS Digital Terrain Model interpretation of the topography and the accurate Q/h water level relationship, the field survey information could be translated in input data for the hydraulic model. With the field survey covering the whole river-bank zone of the stretch, sufficient data of presence and absence were present for the modelling and to calibrate the observations.
As spatial sequences for the softwood forest development in the Common Meuse were described the islands, lateral bars, point bars, levees and flood channels. Temporal sequences of forest development include the germination phase (year 1), the establishing phase (2-3 year) with the development of dense thickets; the survival phase (4-10 year) of thicket to young forest with groups of different age classes; the forest phase (> 10 year) is settled forest on elevated islands and bars in the river bed or higher on the banks and in the floodplain.

Hydraulic model

The hydraulic model SCALDIS (Mwanuzi & De Smedt 1997, Mwanuzi 1998) was used for the hydromorphological modelling. SCALDIS is a 2-dimensional numerical model, based on the finite elements concept. For the model, the river bed is divided in a grid with grid cells of 200 m length and a width varying of 10-100 m. For the riparian zone and the lower floodplain the grid cells had a width of 10 to 20 m. SCALDIS allows the calculation of water level (h), hydraulic radius (R), stream velocity (v) and shear stress (τ) for each grid cell at a given discharge, based on the Manning equation. Figure 4.1 shows the riverbed in profile with the significant water levels for the modelling. For the shear stress at specific elevation z in the riverbed τ_z [N/m^2], the following formula was derived:

$$\tau_z = \frac{\tau}{R}(h - z)$$

The erosive capacity can be calculated by comparison of the shear stress τ_z with the critical shear stress τ_c for bedload transport. The critical shear stress τ_c [N/m^2] is defined in the Shields formula for coarse gravel beds:

$$\tau_c = \theta g(\rho_s - \rho)d_{50}$$

with θ the dimensionless critical shear stress or Shields parameter, ρ_s the sediment density ($\approx$2.65r) [kg/m^3] and d_{50} the median bedload grain size [m].

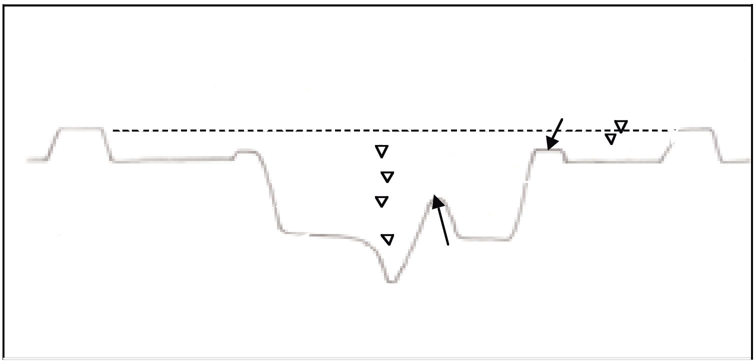

Figure 4.1 Riverbed profile with critical water levels for morphologic development (z3000 is the 3000 m^3/s water level).

Table 4.1 Critical shear stress tc for gravel bedload and armoured layer erosion.

θ	τ_c [N/m²] bedload	τ_c [N/m²] armoured layer	Bedload transport
< 0,03	< 7,3	< 17	No bed movement
0,03 – 0,06	7,3–14,6	17–34	Partial bed movement
> 0,06	> 14,6	> 34	Active bed movement

For the dimensionless critical shear stress three reference values are used for the description of the rate of bed transport (Lisle et al. 2000). The bed layer of the Common Meuse has a gravel fraction with a mean sediment diameter of 15 mm. The armouring of the bed layer results in higher critical shear stresses for bed movement than expected based on the mean perimeter of the bed fraction (Wörman 1992; Raudkivi 1998). For the armoured layer a d_{50}-value of 35 mm was used (Van Manen et al. 1994). Table 4.1 gives the thresholds for the critical shear stress for substrate and armoured layer erosion, calculated with equation (3) for the three q-reference values.

As roughness parameter in the modelling an n-value of 0.030 s/$m^{1/3}$ for the open gravel bed was retained after calibration, for the floodplain an n-value of 0.040 s/$m^{1/3}$. For the resulting forest in the riparian zone a value of 0.10 s/$m^{1/3}$ was used, as it is suggested for dense shrub and forest (Chow 1982). In the description of sedimentation and erosion processes, two additional parameters were derived from the critical shear stress; the entrainment potential as the proportion of actual to critical shear stress and the shear stress gradient, the ratio of change along the stream. The field survey of the distribution and clearing of forest development was analysed for the specific hydraulic conditions of shear stress and bedload with the τ_Z and τ_C formulas.

Results

Spatio-temporal sequences

Erosion-sedimentation rates differ strongly over the area during the survey. Some wooded bars gain 0.5 m a year, while others get washed away completely (table 4.2). The largest bars and islands are associated with larger bed widths and riffles (figures 4.2 and 4.3). No correlation was observed between shear stress values (minima/maxima) and the presence of bars, only shear stress gradients and width/depth ratio's showed correspondence to the position of bars with higher erosion-sedimentation ratio's (figures 4.5 and 4.6). Especially the shear stress gradient over the river stretch accords to the zones with active bar formation and sedimentation/erosion processes. Just downstream gradient peaks active bar formation takes place. High W/d-ratio (> 20 for the Common Meuse) do not always correspond to morphological activity, as at many locations bank protection prohibits bar formation. Nevertheless the criteria for W/d-ratio (> 20) and shear stress gradient (> |0.02|) together give a strong tool to detect and predict morphological activity, necessary for bar formation and riparian forest development processes.
Hydrological conditions differ strongly from year to year and the regeneration of riparian forest consequently occurs in waves, as recruitment and settlement depend on early season conditions and annual peak flows (Van Splunder 1998).
For the surveyed reach, seedling survival was successful at the majority of the bars in 1999 and 2002, not in the years with higher summer peaks 1998, 2000 and 2001 (see figure 4.4). Higher winter peak flows were responsible for the reduction of older phases. The forest recruitment and bar formation can be erased up to the survival phase quite frequently, whereas the developing forest is gradually elevated by sedimentation (figure 4.2, table 4.2).

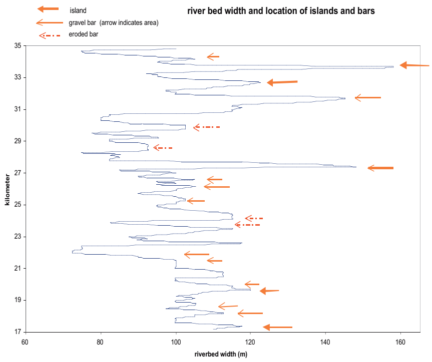

Figure 4.2 Location of bars related to the bed width of the Common Meuse summerbed. The bars that eroded in the winter peak of 1998-1999 are indicated. Numbers of the bars correspond to table 2, arrow length is an indication of bar length (ranging from 100-500 m).

Table 4.2 Bars and islands with field and map survey characteristics. The erosion-sedimentation ratio is the field measured height of erosion (-) or sedimentation (positive) for that period.

				Bed width change [m]		Erosion sedimentation ratio [cm]	
Bank type	Location	Number figure 3	age [y]	'87-'97	'30-'97	'98-'99	'99-'00
Islands	Smeermaas	1	20	0	17	0	0
	Hocht	3	8	5	10	30	-40
	Maaswinkel	13	15	10	10	20	5
	Meers2	14	30	0	50	0	0
Lateral bars	Borgharen	2	4	0	5	20	10
	Hocht	3	3	0	0	20	-10
	Herbricht2	5	4	0	0	20	0
	Herbricht3	5	5	0	0	-20	5
	Geulle	6	2	2	2	40	10
	Kotem1	7	8	10	10	30	0
	Kotem2	8	3	10	10	0	-10
	Maasband	15	5	10	10	-5	0
Point bars	Kotem-Hal2	11	5	0	10	-90	-20
	Itterse Weert1	4	4	0	0	10	0
	Itterse Weert2	4	5	0	0	30	0
	Herbricht1	5	5	0	0	40	-10
	Kotem-Hal1	10	8	-10	0	-20	-10
	Meers1	12	7	10	10	30	0
	Elsloo	9	5	50	50	30	0

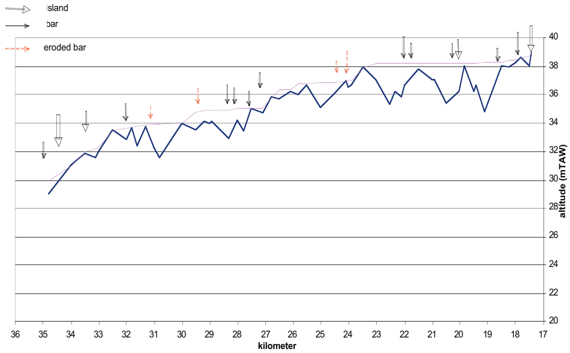

Figure 4.3 Longitudinal profile of summer bed and low flow (minimal discharge of 10m^3/s) and the location of bars and islands. Length of the arrow is an indication of the bar length (between 100-500 m).

Table 4.3 Spreading of the temporal sequences over the bar types in the survey (number of bars with sequence present).

		seedlings	establishing	thicket (survival)	woodland
Point bar (7)	1998	7	2	1	0
	1999	5	1	0	0
	2000	2	1	0	0
	2001	2	2	1	0
	2002	4	2	1	0
Lateral bar/ island (12)	1998	12	9	6	2
	1999	9	7	5	3
	2000	3	7	5	3
	2001	2	7	7	3
	2002	5	7	7	3

Important observations are the small summer peaks of 1998, 2000 and 2001 which disrupted the longer low flow period for germination on most of the bars, resulting in less seedling sequences in table 3 (numbers for 1998 were collected before September peak flow). The high winter peaks for the surveyed period resulted in a strong reduction of temporal sequences starting from a rich situation in 1998 after a few years with less pronounced winter peaks.

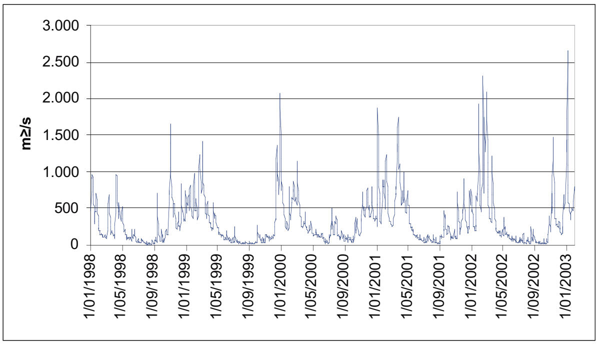

Figure 4.4 Average daily discharge values for the Common Meuse over the survey period.

The critical boundary conditions for the germination phase were derived from the observation of summer peak effect on young seedling growth, allowing the identification of the critical shear stress for seedlings. A 350 m^3/s peak washed away all seedlings on the bars below the z100-line, the line corresponding with the 100 m^3/s discharge level. The mean values of shear stress tz at the critical discharge levels over the surveyed lateral and point bars were derived from the model, and retained as critical shear stress for the first temporal sequence (table 4.4). The calibration of the stand conditions under these hydraulic stresses, gives by recalculation of q values between 0.025-0.054 stating the boundary condition for bedload movement (table 4.1). This gives a validation of the model assumptions and shows the mobilisation of the substrate and the resulting derouting/abrasion as the critical parameter in the forest development.

Table 4.4 Critical shear stress ranges for the forest development phases.

Phase	Shear stress range [N/m²]		Measured at discharge [m³/s]	Recurrence of this discharge in 10 year period	Measured at elevation
Germination	0.3 - 4.5	point bar			
	0.3 - 6	lateral bar	350	18.3	z100
Establishing	9 - 11.5	point bar			
	11.5 - 13	lateral bar	800	20.6	z250-350
Survival	13 - 17	point bar			
	13 - 22	lateral bar	1500	5	z250-350
Riparian	< 30	point bar			
forest	< 33	lateral bar	3000	0.2	z250-800
Germination	0.3 – 0.76	flood channel			
floodplain	0.76 – 2	levees	1500	5	z800
Settlement	2 – 4.5	flood channel			
floodplain	4.5 – 11.5	levees	2000	1	z800
Floodplain	< 10	flood channel			
forest	< 13	levees	3000	0.2	z800-1500

For the further temporal sequences, the critical water levels and shear stresses were derived from the observation of abrasion of shrubs and forests at certain discharges, by calculating the critical shear stresses (lower and higher boundary conditions and mean values) at the specific locations. These critical boundaries result in the definition of ranges for the spatio-temporal sequences (table 4.4).

Future germination and growth

The hydraulic modelling gives a valuable tool in the prediction of riparian forest development, with the determination of morphologically active zones and spatio-temporal sequences of development.
The riverbed zones with high shear stress gradients do not alter/change drastically in the restoration project from the present situation, as there will be no measures in the gravel bed. With the bank lowering measures, a few peaks become a little lower and a few are more pronounced. The W/d-ratio criterion is reached in 80% of the stretch after restoration, so for the future development, the shear stress gradient is more discriminating for the possibilities of bar formation. Figure 4.7 shows the critical shear stress ranges for riparian forest development at a two year recurrent peak flow in the future situation after restoration measures, with delineation

of the morphologically active zones. In Figure 9 the result of the modelling of forest development sequences is shown for the future situation.

Discussion

Many authors point out critical conditions of hydro regime and reversible processes and patterns for riparian forest development (Amoros et al. 1987, Auble et al. 1994, Girel et al. 2001, Hughes et al. 2001). The integration of hydromorphic and biotic sequences for modelling and prediction of forest restoration in the river system was yet never really achieved.

The hydromorphic regime was determined as driving force for the allocation of the forest development stages. Hydraulic stress in the germination phase was detected in the delineation of bedload movement. For the establishing phase, mortality was linked with the execution of excessive forces on the trees, resulting in abrasion. The hydrograph of the research period (figure 4.4) explains this criterion. Flood duration of individual peaks never exceeds critical periods of 2 weeks, nor does annual flood duration attain critical levels of >100 days as critical range for softwoods of Salix and Populus. Therefore the morphodynamics act as sole criterion in riparian forest development for this river stretch. This observation contrasts to other river surveys where flood duration was attributed equal explanatory value as morphodynamics (Naiman et al. 1997, Van Splunder 1998, Friedman & Auble 1999).

For an adequate prediction of forest development for flow resistance matter, the distinction of spatio-temporal sequences in the modelling is primordial. Especially the distinction of a geomorphic and biotic component in the development is an essential step in the elaborated method. The better allocation of forest development, improves the quality of flow resistance modelling. The with this approach obtained outcome showed a significant water level decrease (average of 9 cm over the whole reach) at normative discharge, in comparison to the generally used modelling approach with randomly generated forest patches. The discussion on forest development and flow resistance came to a better consent with this approved model application.

From the presented analysis we can derive some guiding principles for restoration approaches. The geomorphic component needs a management strategy at river reach scale allowing lateral dynamics of free eroding banks and shifts in channels, providing for sufficient sediment supply and morphological activity in the river bed. For the biotic component, the provision of natural flow conditions, with necessary dynamics to create and control reforestation prevails. At the reach scale, the provision of space and freedom for the river is crucial to maintain and create the spatio-temporal sequences in a viable way. Especially a detailed target setting at the site level risks endangering the goals of riparian forest restoration. For a sustainable forest development, all the spatio-temporal sequences need to be present in a viable way to provide for a sufficient seed rain, gene pool and habitat for specialist species at reach level. Recent investigations showed the aspects of gene flow and connectivity as crucial aspects in the restoration of riparian forests of Populus nigra, with emphasis on the problematic situation of fragmentation and isolated stands in the lowland reaches (Imbert & Lefèvre 2003). Black poplar was identified as recruitment-limited rather than dispersal-limited in the lower river reaches where pioneer habitats are limited. The problem of fragmentation was documented for the Common Meuse river forest (Van Looy et al. 2003), as were the genetic problems for Populus nigra, where preservation of present stands and even reintroduction proved necessary to restore the species gene pool (Vanden Broeck et al. 2004). For this reintroduction a variety of locations was selected, well connected to the river and as close as possible to the locations indicated as potential sites for riparian forest development in this modeling. So, conclusions from the presented approach can be drawn towards a dynamic approach of restoration efforts.

Acknowledgment

This study was funded by the Flemish government nature development research programme VLINA (VLINA0012).

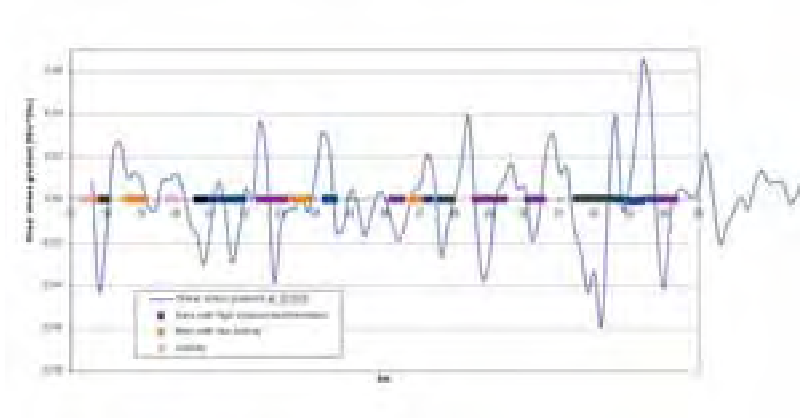

Figure 4.5 Shear stress gradient at bankfull discharge (1500 m^3/s).

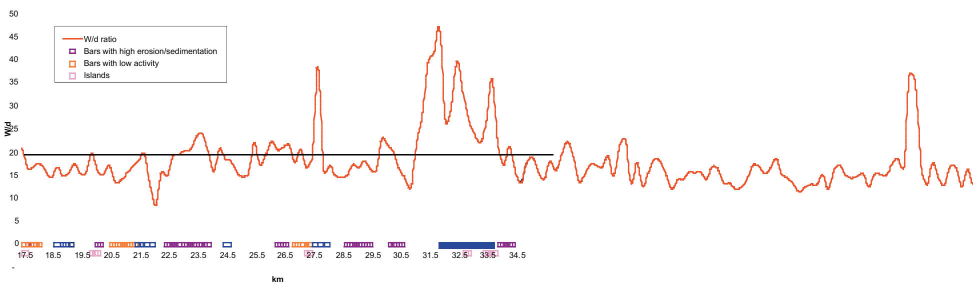

Figure 4.6 Width/depth ratio over the river stretch with the location of the bars and islands.

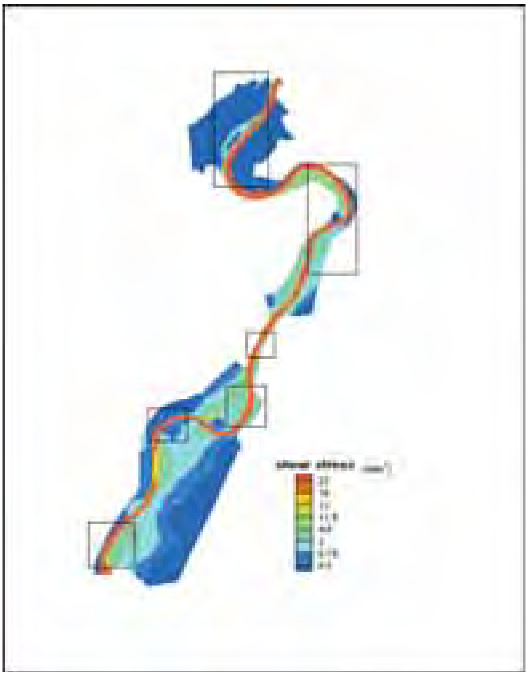

Figure 4.7 Shear stresses at peak discharges (2000 m^3/s) for the future situation after rehabilitation measures of river stretch km17.4-34.8.The boxes delineate the morphological active zones of the riverbed as determined in figure 6-7.

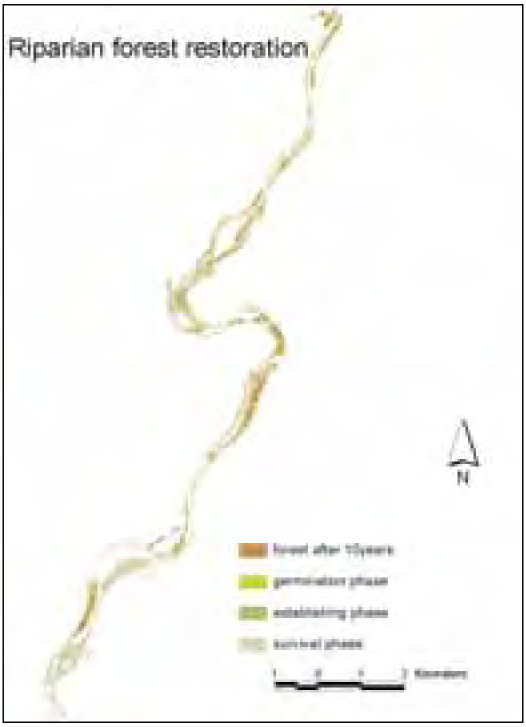

Figure 4.8 Model prediction of forest development after restoration measures for the study area. The temporal sequences are presented, with the summary 10 year forest as overlap of the three sequences.

IV.2

A conservation paradox for river corridor plants.

Kris VAN LOOY and Patrick MEIRE

Submitted to Biological Conservation

Abstract

We investigated grassland composition and diversity aspects for the alluvial plain of the River Meuse, one of the larger Northwest European streams, with special emphasis on the river corridor plants of dry river grasslands. In order to define a conservation strategy for the river corridor plants we examined isolation and fragmentation aspects and the role of flooding. A mapping and sampling of vegetation and soil conditions over the alluvial plain was executed, together with a recruitment analysis for the rare species of dry river grasslands. The central question for the study was whether preservation of relicts is a sufficient means to preserve riparian diversity. In the DCA ordination the rare river corridor plants were clearly restricted to the pioneer dry river grasslands of gravel or sandy deposits further from the river. A significant isolation of the river corridor plant relicts was revealed. As for the cause of this isolation, our analysis indicated recruitment limitation to be the major threat for survival of most of the river corridor plants. The recovery of populations depends strongly on flood contact and recruitment potential in the creation of new habitat. The withdrawal of the hypothesis that conservation outside the river dynamic influence is a necessity, shows that the construction of conservation and rehabilitation strategies for species at risk needs a good knowledge of key processes that determine the population dynamics at the regional scale. For the investigated River Meuse reach, the flood dynamics proved an essential habitat creation process, strongly determining population dynamic strategies and restoration potentials at the reach scale.

Introduction

Riparian zones are considered hot spots of species diversity (Gregory et al., 1991; Ward, 1998). Understanding the mechanisms which generate the plant species diversity in the riparian landscape is a challenge in the attempts to preserve these diversity hot spots (Zwick, 1992; Tockner et al., 1999).
The heterogeneity in abiotic conditions and the presence of strong gradients over the river corridor, together with the function as migration route, is put forward as explanation for the observed biodiversity (Nilsson et al., 1989; Petts & Bradley, 1997; Pollock et al., 1998; Ward et al., 1999). Also, a strong local-regional connection in species pools is thought to exist in river landscapes and

explains the high diversity (Naiman e.a., 1993, Mouw & Alaback, 2003).
A group of Central European plant species expanded their distribution range to the north along the large river's corridors. These river corridor species took advantage of the wide floodplains of the large rivers to reach the lowlands of Northwest Europe. The river corridor plants are a highly appreciated nature conservation asset of the River Meuse, the same as for most Northwest European streams (Burkart, 2001; Donath et al., 2003; Jongman, 1992). Emphasizing on this group and the factors limiting its distribution, is a possible way to develop conservation strategies for the floodplain diversity.

Human-induced changes to flow regime, flood contact and groundwater level cause a deterioration of the diversity of the riparian landscape (Petts, 1996; Ward, 1998). Together with the intensification of agricultural practices in the alluvial plain, this makes the dry river grasslands very fragmented and the characteristic river corridor species highly threatened in the present situation. Disruption from flood contact of large parts of the alluvial plain by the construction of winter dikes, is a further threat in this fragmentation problem (Leyer, 2005), as we already highlighted for floodplain forest diversity (Van Looy et al., 2003) for the highly regulated River Meuse.

Central question for our research was 'Is the preservation of relicts a sufficient mean to preserve riparian diversity'. At present, conservation efforts focus on the protection of relicts of these dry river grassland communities, in the designation of special protection zones within the Pan-European NATURA2000 network. In further attempts to define conservation strategies for these communities, river concepts like the shifting mosaics and the patch dynamics concept (Pickett and White, 1985; BarratSegretain and Amoros, 1996; Petts, 1996; Petts and Bradley, 1997) provide useful frameworks for the problem description and definition of spatial and management guidelines. With the described alterations to fluvial functioning by regulation, resulting in habitat deterioration and fragmentation, the dynamics of patches and populations received strong emphasis in our study. Starting from a mapping and diversity analysis of the grasslands in the alluvial plain of the River Meuse, we focussed further on specific habitat conditions and distribution of the dry river grasslands and their river corridor plant species. River corridor plants include a high proportion of threatened plant species with small populations, dispersed over a restricted number of patches alongside the river (Baumgärtel &

Zehm, 1999; Burkart, 2001; Bisschof, 2002; Jäkäläniemi et al., 2005). In order to preserve them, and in order to understand the mechanisms generating their distribution patterns, more has to be known about their population biology and dynamics (Malanson, 1993; Burkart, 2001; Lowe, 2002). We distinguished population dynamic strategies of the species at risk in order to determine applicable conceptual frameworks for the further planning of the restoration programme for this river reach. The strategies of the river corridor plants in the dry river grasslands are often remnant or metapopulation strategies, very sensitive to local extinctions as a consequence of their strong habitat selectivity. The paradox for conservation lies in their need for flooding processes for habitat creation and seed dispersal and on the other hand their sensitivity to flooding and the fact that relicts can be destroyed by erosion-sedimentation processes. The preservation of relicts outside the dynamic flooding zone is suggested to be a necessity for a sustainable protection of these species by several authors (Jongman, 1992; Hegland et al.., 2001; Donath et al., 2003; Eck et al., 2004; Lindborg and Eriksson, 2004). To test the hypothesis of relict conservation, a recruitment analysis was carried out for rare species with known distribution over the studied reach, and a population strategy testing for these species. To reinforce and argument their conservation, we investigated whether the river corridor plants are good indicators for well developed grassland communities and the overall riparian diversity.

Studied River Meuse reach

The study area is the Flemish side of the alluvial plain of the middle course section of the River Meuse between Maastricht and Maaseik (30 km) on the border between Belgium and the Netherlands, the so-called Common Meuse. Discharge levels for the Common Meuse range from 10 m^3/s during dry periods to 3,000 m^3/s in periods of heavy rainfall in the catchment area. The unregulated Common Meuse stretch is a typical gravel river with a strong longitudinal gradient (0.45 m/km). The Common Meuse valley consists of a gravel underground with a loamy alluvial cover. Local irregularities of levees and dikes are covered with more sandy sediments, the same as for dynamic overbank sedimentation zones. The floodplain traditionally was agriculturally used as meadows. Large parts of the alluvial plain have been excavated for gravel mining, leaving large gravel pits or lowered floodplain zones. The degradation of the floodplain natural heritage was the reason to start a river restoration programme and to start local pilot projects, mostly in abandoned gravel mining locations. The large-scale river restoration project is defined in a master plan for the alluvial plain (Pedroli et al. 2002). The concept of the restoration project is to restore hydrodynamics and morphodynamics and related ecological characteristics in a broadened river channel and in re-established secondary channels and backwaters. Planned measures comprise bed widening, bank lowering and side channel reconnection in a comprehensive approach for the river reach.

Sampling

The vegetation survey of the Meuse alluvial plain consisted of a vegetation mapping with sampling for every recorded patch. A minimum of 500 m? was chosen for the delineation of homogeneous vegetation patches in the field. For the mapping a topographic map basis was used. The parcels in intensive agricultural use were all integrated as rectangular patches in the map, for the natural managed areas, more irregular forms of patches arise. The 196 patches of grasslands under natural or extensive management were sampled in 1999 using 1 x 1m relevees. For the relevees, the Braun-Blanquet method of 1x1m quadrate sampling was used, as it was recorded useful for a biodiversity analysis at different scales (Pollock et al. 1998). All species within the sample plots were recorded. The grasslands cover a range from open pioneer to dense, tall vegetations. They were classified in 9 types according to management, elevation and river dynamics (Table 4.5). These types were assigned to a corresponding phytosociological association or order according to Schaminée et al. (1998).

Table 4.5 Classification of grassland patch types in the Meuse alluvial plain , with annotated phytosociological community (Schaminée et al. 1998).

Agricultural practice	B1 hayfields	Arrhenatherion elatioris
	B2 pastures	Cynosurion cristatus
	B3 fertilised meadows	Poö-lolietum perenne
Natural management		
Lower floodplain meadows	F7 long inundated meadows	Lolio-potentillion anserinae
	F9 floodplain meadows	Alopecurion pratensis
Higher floodplain meadows	L1 dry river grasslands	Medicagini-avenetum pubescens
	L4 xeric grasslands of open sand	Thero-airion caryophyllea
Overbank sedimentation zones	A1 gravel overbank sedimentation	Alysso-sedion albi
	A2 sand overbank sedimentation	Sedo-thymetum pulegioides

Environmental variables were gathered in the field survey or derived from available digital data on the flooding and from the mapping in GIS.

Environmental variables were gathered in the field survey or derived from available digital data on the flooding and from the mapping in GIS. Flooding frequency of the samples ranges from more than once a year to less than once within a decade. It was derived from the two-dimensional hydraulic model developed for the restoration model and based on a high resolution DEM of the alluvial plain (See Van Looy et al., 2005). The frequencies were divided in flood frequency classes (>1/year, 1/year, 1/2-5year, 1/5-10year, < 1/10year). We also determined for each plot the distance in bird's-eye view to the river channel (m.). Isolation was recorded in categories, measured as the distance to the nearest same patch type, distances ranked in categories (1: <50m, 2: 50-500m, 3: 500-2000m, 4: >2000m). Management was classified as extensive meadows with haying and/or pastures (2), natural grazing (1) and no management (0). Soil humidity classes are wet (3), periodically wet with high fluctuation (2), moderately dry (1), extremely dry (0). The organic matter in the topsoil layer was categorised as a thick humus layer (2), present (1), absent (0). Soil texture in each sample plot was manually analysed and categorized in 9 classes, from clay (1), silt (2), loam (3), sandy loam (4), loamy sand (5), clayey sand (6), sand (7), gravel-sand (8) to coarse gravel (9). This texture classification was checked for 50 plots with a soil sample laboratory analysis for texture

(laser diffraction), acidity with Metrohm (titration, pH-carrousel), organic matter (Moffeloven destruction analysis) and conductivity (EC measured with conductance meter and translated to soil salinity).
Two consecutive exceptional floods in 1993 and 1995 showed the highest ever recorded peak levels. After these extreme peak events, a survey was done for the overbank sedimentation zones. Substrate texture was determined, floristic inventories carried out in the summer period and the zones were mapped. As these newly created habitats proved very important for the recruitment of river corridor species, this survey was repeated after the peak discharges of 2000 and 2002.

Ordination and diversity analysis

In a first stage we performed a data exploratory Detrended Correspondence Analysis (DCA) using the CANOCO 4.0 software (Gauch, 1982; ter Braak and Smilauer, 1997). Only species occurring in more than one plot were used for this analysis. In order to identify the abiotic drivers of the species composition gradients, DCA sample scores of the 196 sample plots were related with flood frequency class, isolation, soil humidity, organic matter and management using a one way Analysis of Variance (ANOVA) and with distance to river, flooding frequency and soil texture using a Spearman rank correlation coefficient.
For the diversity analysis we selected the rare species (≤ 5 plots) in the data set. Then we related species richness and diversity of the plots with rare species to the environmental parameters. We used a one way ANOVA. All statistical analyses were performed with Statistica (StatSoft Inc., 2001).

Population strategies

Freckleton & Watkinson (2002) defined population dynamic strategies explaining spatial dynamics of plants on a regional scale. They proposed a classification of large-scale spatial dynamics based on the relative importance of regional and local dynamics for the persistence of plant populations.
To classify the species dynamics in the Meuse river system, the Freckleton & Watkinson typology was translated into a scheme of species and patch criteria (Table 4.6). The strategies were appointed based on species frequency and abundance in the plot-species matrix and the vegetation mapping. The main distinction is between regional and local populations. In terms of the application of metapop-

ulation theory, regional populations are relying on colonization from upstream populations.

Table 4.6 Classification criteria for population dynamics types

Population dynamics type	Source population, immigration	Population frequency	Abundance within patch	Patch type selectivity	Patch frequency	Occupation of suitable habitat	Patch dynamics	Patch size, isolation
Meta-population	Upstream	Rare-occasional	Rare-occasional	high	frequent	partially	low	small, dispersed
Source-sink	Upstream	Rare-occasional	Rare-frequent	low	frequent	low	high	dispersed
Remnant population	Local	Rare-occasional	Rare-occasional	high	rare-occasional	partially	low-high	small, isolated
Shifting cloud	Local	Rare-frequent	Rare-frequent	low	frequent	low	high	-
Patchy population	Local	Rare-frequent	Occasional-frequent	high	occasional	high	medium-high	small, dispersed
Extended local population	Local	Frequent	Frequent	low-high	frequent	high	low	large

We classified the rare species (≤ 5 plots) in this population dynamics typology following Table 4.6. An indication to the dependence of upstream populations is the absence of strong local populations and the presence of occupied patches widespread along the river's axis. Species with only few and small local populations (abundance criterion: rare to occasional Tansley abundance values in the relevees and isolation criterion) were classified depending on colonization by upstream populations.

The second criterion number of populations was also used as criterion for the diversity analysis and is for our selection of course low (rare, <= 5, occasional <20). The extended local populations are out of the scope of our analysis of rare species as they are frequently present populations. For the abundance within the patches the Tansley cover in the relevees is used. Patch type selectivity refers to the determined grassland types of the mapping. Selectivity is low for generalist species present in >2 types. Patch frequency refers to the presence of the grassland type over the alluvial plain mapping (rare <=5, occasional <20). For the grassland types the species occur in, the occupation of suitable habitat describes the share of the patches of these types where the species are

present (low <10%, partially 10-50%). The patch dynamics for the specific situation of the floodplain and the river corridor plant strategies, are related to the river dynamics; high dynamic patches are regularly flooded with high river morphodynamics, patches with medium dynamics are regularly flooded and low dynamics means irregular flooding (< once/2year).
For the patch size criterion, small patches are in average <0,5ha. The isolation is measured in the distance between occupied patches. Isolated patches are >2000m apart.

Results

The mapping shows that over 50% of the alluvial plain is in intensive agricultural use (Figure 4.9). The dry river grasslands and the pioneer stage of overbank gravel and sand depositions, take only 4% of the alluvial plain. Natural riparian landscape units like sand-gravel bars, pioneer vegetation and overbank sediment zones have extremely low values, together they take only 1% of the surface.
The alluvial plain consists mainly of larger patches in agricultural use (arable land patches mean area 4,2ha). Some nature reserves and riverbanks show smaller vegetation patches.

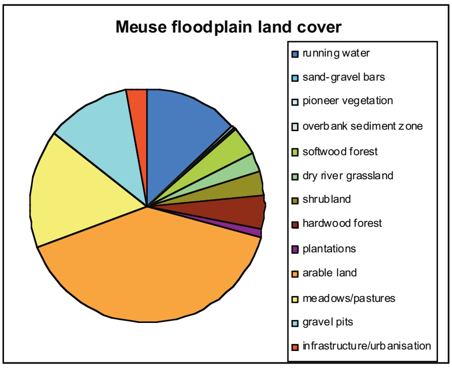

Figure 4.9 Pie chart of land cover units over the Common Meuse alluvial plain.

The vegetation survey yielded 329 species, 226 of them were present in more than 1 plot and retained for the ordination analysis. In the diversity analysis were entered the 46 species occurring in 2-5 plots (Table in annex). Species richness and patch area were plotted for the grassland types (Figure 3.5), indicating that the dry river grasslands (types A1, A2, L1 and L4) are the richest communities over the alluvial plain with the smallest patches. This indication is even stronger in the species-area plot (Figure 4.10), showing there is a strong concentration of rare species in the smallest and most species-rich patches.

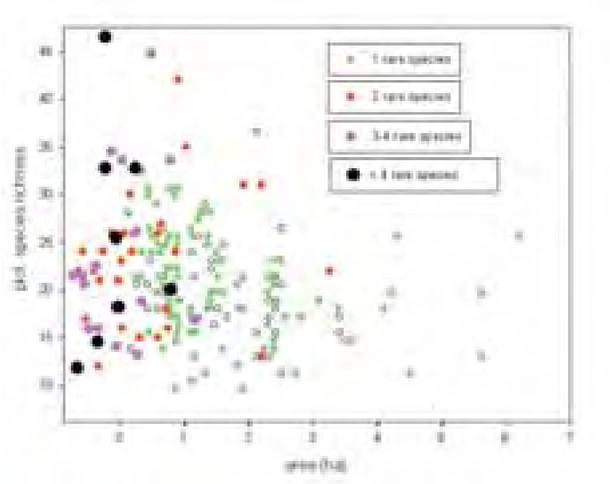

Figure 4.10 Species-area relationship for the sampling. Plots with no rare species are open circles, the larger dots indicate plots with more rare species.

More than half of the rare species are river corridor species (27/46) of the northern Central European streams (Burkart 2001). When we include the very rare species, in only one plot represented (which we omitted from the analysis) a majority of the Meuse river corridor plants is in this category.

Ordination results

The first three axes of the DCA explained a cumulative percentage of the variance of the species data of 19%, with gradient lengths > 6, expressing the heterogeneous character of the vegetation in the sample plots. The first DCA-axis shows a wet-dry gradient. There are only a few wet meadow patches present. For the most part of the alluvial plain summer groundwater levels are about 3-5 meter below ground as a consequence of the river bed incision of the last century. The significant relation between DCA1 sample scores and flooding frequency and soil characteristics of texture, soil humidity and organic matter expresses the river influence in the floodplain environmental conditions. DCA2 sample scores showed significant covariance with soil parameters and management conditions and also with isolation and distance to the river (Table 4.7). For this axis the management and soil texture are the most explanatory abiotic characteristics, showing a gradient of densely vegetated floodplain meadows to open, sandy pioneer grasslands. Hayfield species and nutriphilous species have low values, whereas sand- and calcareous, xerophilic species have high DCA2 values. For the third axis, flooding frequency and distance to the river show strong covariance.

Table 4.7 Covariance test results for stand conditions and ordination axes.

	distance river	texture	flood frequency	isolation	humidity	organic matter	flood frequency	class management
DCA1	0,18*	0,33**	-0,4**	1,11	37,5**	32,9**	32,6**	4,9*
DCA2	-0,27**	0,45**	0,04	8,3**	6,7**	17,5**	0,5	10,1**
DCA3	0,38**	0,02	-0,38**	0,42	0,81	0,74	6,7**	1,2
DCA4	0,2*	0,089	0,07	2,3	2,5	0,8	1,2	0,99
Spp richness	0,14	0,16	-0,22*	2,1	3	5,22*	3,08*	0,9

Distance to river, substrate texture and flood frequency are Spearman rank correlation coefficients, for the other variables ANOVA F-values are given. **: significant correlation (p< 0,001), *: little significant (0,001< p < 0,01)

In the ordination the rare river corridor plants are clearly grouped together with the pioneer grasslands of the dry river grasslands on gravel or sandy deposits further from the river and irregularly flooded (Figure 4.11). The rare species of the dry river grasslands show strong correlation with the third axis (Rare species-DCA3 z:5.74, p<0.001, Figure 4.12), indicating the isolated position of the river corridor plant relicts situated farther from the river and seldom flooded.

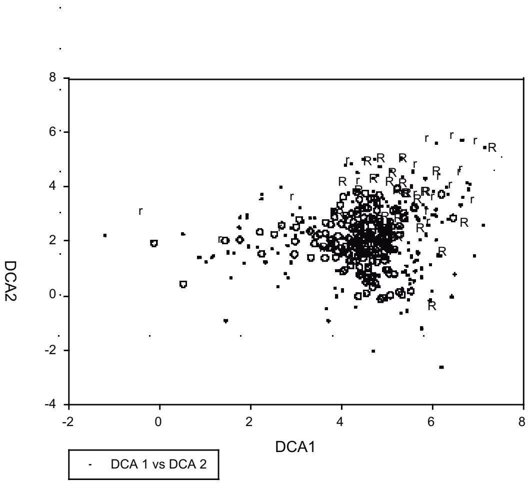

Figure 4.11 Plots-species biplot over the first two DCA axes; with circles: plots, dots: species and r: rare species, R: rare river corridor species

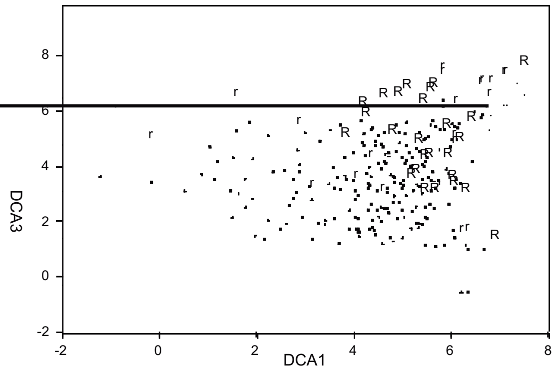

Figure 4.12 Species biplot over the DCA axes 1 and 3; with r: rare species, R: rare river corridor species

Diversity analysis

The number of rare species shows significant covariance with the species richness of the plot (F: 3.6, $p<0.001$). This marks the rare species as good indicator species for the well developed grassland patches of the alluvial plain. Higher rates of rare species are restricted to the plots with gravel and sand soils (F:4.6, $p<0.001$). This shows their faith to the dry river grasslands, as the gravel-sandy soils are only present at levees and overbank sedimentation plots, the stand conditions for the dry river grasslands.

The strongest covariance is between the rare species (F: 20.6, $p<0.001$) and the isolation. Together with the observed restriction to the smallest patches, this indicates that fragmentation of habitat is a major threat for the rare (river corridor) species. The number of rare species clearly increases with the degree of isolation (Figure 3.9). The graph shows there's a group of isolated relict sites, harbouring a list of specific rare species. The observed isolation does not necessary imply that disconnection of river contact is the problem. Fragmentation of habitat, by loss of habitat through land use changes can cause isolation as well. Species truly suffering from isolation have lost their dispersal abilities and colonization potential. The inventory of newly created habitat patches after the floods of 1993, 1995, 2001 and 2002 proves the opposite. The rare species of the dry river grasslands show a high colonization potential (Table S4 in annex). In 21 newly created patches (ranging from 50m? - 2ha) of overbank gravel-sand deposition over this investigated period, most rare species show recruitment of the new habitat. The recruitment of river corridor plants of the dry river grasslands was only successful over the investigation period (between 1993 and 2002) in the extensively managed areas, as the sediment zones under intensive agricultural use are reworked (evened/ploughed) after each flood event with destruction of the newly created habitat.

The population dynamic strategies were appointed (Table S4 in annex) based on species frequency and abundance in the plot-species matrix and mapping. We derived a recruitment rate for the different species strategies, by analyzing the recruitment over the strategy groups (Table 4.8). Mean values for the groups give a good approximation of a recruitment rate for the metapopulation, remnant and patchy population dynamic strategies. Especially the patchy population strategists are very successful in colonizing suitable habitat patches. The recruitment of newly created habitat is mostly an immediate process (seeds provided with flooding) covering most of the rare species in the immediate surroundings.

Table 4.8 Surveyed recruitment of river corridor plants within the population strategies.

Strategy	**Remnant**	**Patchy**	**Metapopulation**	Source sink	Shifting cloud
Number of species	12	28	8	1	2
Recruitment	13	116	11	4	12
Median + SD	1 + 1	4 + 1.2	1 + 0.8	-	-
Mean	1.08	4.14	1.38	4	6
Recruitment rate	1,1	4,1	1,4	4	6

Species showing no recruitment do have a dispersal limitation, this can be due to disruption from flooding or to a regeneration limitation in unfavourable relict conditions. For some remnant species, e.g. Potentilla argentea and Sedum telephium, the contact with the river is lacking, and no recruitment was observed. To this list can also be added a list of species restricted to one plot in our analysis: Potentilla neumanniana, Carex caryophyllea, Briza media, species that are disconnected from river contact by the construction of dikes. Other remnant species like Eryngium campestre, Ononis spinosa, Plantago media and Tragopogon pratensis show only limited recruitment due to a lack of regerenation under actual unfavourable stand conditions of changed agricultural practices.

Discussion

Burkart (2001) described the distribution pattern of the river corridor plants of the large northern Central European rivers (Vistula, Elbe, Oder and Weser) of which 48 of the 129 are also present in the Meuse corridor. Some of the here identified rare river corridor plants (Sedum sexangulare, Vulpia myuros and Trifolium campestre), are also distinguished by Baumgärtel and Zehm (1999) as characteristic species of the Rhine system sandy deposit mosaics. These authors tried to derive explanations for their threatened status and potential restoration guidelines from their remarkable distribution. The ordination and diversity analysis together with the recruitment and population strategy assessment proved a successful method to analyse the postulated conservation paradox, as they revealed the patterns and threats in the actual distribution of river corridor plants along the Meuse. There was a clear segregation of rare (river corridor) species in the ordination, showing the specific status of the dry river grasslands rich in river corridor plants. The rare river corridor plants seem good indicators of diversity and fragmentation aspects at river reach scale, as the rare species correlated significantly to species richness and isolation. The identified correlation with isolation of the dry river grasslands, proves their threatened status and need for restoration.

The dispersal limitation of the river corridor plants in remnant populations

along the Rhine was recorded as most important limiting factor for restoration success (Donath et al., 2003). Therefore the preservation of the relicts was put forward as the most crucial together with the provision of sufficient habitat adjacent to these sites. Hegland et al. (2001) came to the same conclusion based on the same method of population analysis for a river corridor plant (Salvia pratensis) along the River Waal in the Netherlands. This preservation strategy was also put forward by other authors (Jongman, 1992; Eck et al., 2004). Bischoff (2002) observed strong dispersal limitation in a floodplain with little flood dynamics (very low flow velocity). These observations contrast strongly with our observations for the River Meuse of strong flood related dispersal capacity and recolonization potential for most river corridor species. Our analysis indicated recruitment limitation to be the major threat for survival of most river corridor plants of the dry river grasslands. Wolfert et al. (2002), Boedeltje et al. (2004) and Baumgärtel and Zehm (1999) also pointed at the necessary stand dynamics and flood pulse based on the study of abiotic conditions of dry river grasslands with characteristic river corridor plants. Eck et al. (2005) documented also recruitment limitation along disturbance gradients as structuring distribution patterns in river floodplains.
In our Meuse dataset, the communities with rare river corridor plants were restricted to overbank deposition zones, linked to the periodic habitat creating process of overbank deposition of gravel and sand sediments. The recruitment analysis showed the potential to colonize newly created habitat for most of the threatened species. Recruitment limitation proves the major cause of threat for most of the river corridor species. Species showing limited recruitment, indicated a dispersal limitation due to disconnection of flooding contact. As we were interested in the impact of fragmentation of habitats to the conservation strategy, in the recruitment analysis, we revealed the impact of the recent dike construction to the distracted relicts, as we already documented the strong impact of this disconnection to floodplain forest diversity (Van Looy et al., 2004). Other authors also described the importance of water dispersal (Johannson et al., 1996; Nilsson et al., 1989; Andersson et al., 2000b) and the barrier effects of dikes (Andersson et al., 2000a; Leyer, 2005) for floodplain grassland species.

But furthermore, we revealed the necessity of dynamics for the conservation of these species, as we observed that only dynamic habitats, with species in more dynamic strategies, show potentials to recruitment and restoration in general. For most species, the population strategy assessment explained these patterns. The Freckleton and Watkinson population strategy classification differentiates between spatial scales of population structure, enabling conclusions towards aspects of the river continuity. Main distinction is between regional and local populations, for the river system regional populations in terms of the application of metapopulation theory are relying on colonization from upstream populations. The species were assigned to one of these strategies without the evidence of a lengthy population study and no reference is made to current discussion on the distinction of metapopulations in non-continuous habitats and the evidence for extinctions and discrete habitat patch use (Gouyon et al., 1987; Ouborg, 1993; Eriksson, 1996; Freckleton & Watkinson, 2003). Nevertheless this generalized strategy interpretation offers interesting opportunities to analyze aspects of species dispersal at a regional scale (Freckleton & Watkinson, 2002).

Population dynamic strategies of the species at risk, explaining the regional persistence and patterns in populations, can be guiding in the delineation of biodiversity conservation strategies (Miles, 1979; Tilman, 1988; Tabacchi et al. 1996; Hansen et al., 1999; Freckleton and Watkinson, 2003). The Freckleton & Watkinson typology provides a framework for the distinction of regional components of population dynamics, by integrating the key processes that determine the population dynamics (Eriksson, 1996; Hanski & Gilpin, 1997). It is a useful tool in determining how populations persist at the regional scale and important for the construction of conservation and rehabilitation strategies for species at risk (Freckleton & Watkinson, 2003; Jäkäläniemi et al., 2005). Population structure and spatial dynamics are recorded in many studies for their conservation implications for riparian vegetation communities and endangered species (Van Treuren et al., 1993; Brys et al., 2003; Tero et al., 2003).
The population strategies assessment allowed the evaluation of the isolation threat risks, whereas we can conclude to the general importance of the relict conservation, as well as include conclusions of this analysis in the restoration programme. Patchy populations of species colonizing each newly generated habitat near to even far downstream, show an optimal recruitment as was observed for some extremely rare river corridor species, like Medicago falcata, Anthyllis vulneraria and Salvia pratensis. For the metapopulation and remnant

population strategists, upstream sources are of major importance and the management and conservation of present relicts is primordial, but these are by far a minority. Survival of the metapopulation and remnant species' populations is critical under the identified threats. The opportunities for these species lie in the ability of upstream populations to recolonize the Common Meuse reach, but have to be regarded in the scope of declining populations at river basin scale. For some of them, the optimisation of management practices of the relicts, might be sufficient to strengthen the local population and its dispersal and recruitment ability.

Restoration projects in general aim at mitigating the effects of regulation works by rehabilitating geomorphological processes, to promote the recovery of degraded biota and the floodplain benefits from the river (Tockner and Schiemer, 1997). However, the hydrological, geomorphological and biological heterogeneity and variability of river-floodplain systems, both temporally and spatially, can complicate the restoration schemes (Amoros et al., 1987). The role of gradients in hydrological and soil nutrient conditions, determined by the flood regime, together with aspects of spatial and temporal disturbance and connectivity patterns in the river system was already documented for the floodplain grassland biodiversity conservation of the River Meuse (Grévilliot et al., 1999; Grévilliot and Muller, 2002; Vécrin et al. 2002; Geilen et al., 2004). Although the river corridor plants benefit from floods, flooding also bears the risk of local destruction of habitat and populations. So, enhancing flooding can seem a sure restoration strategy for threatened metapopulation strategists, depending on upstream sources for new recruitment. True remnant populations on the other hand first have to be strengthened and/or expanded locally before dynamic restoration can be the best option. The preservation of relicts is also important as upstream populations for many species are strongly decreased and threatened as well as local populations.

As the process of habitat creation does not annually occur, the spatial conclusions of the recruitment analysis also need a temporal interpretation. Therefore we can project the habitat creating process over a broader range of potential locations, with of course the restriction that existing habitat will be put back in succession, as local stands will be overdeposited or eroded, or become temporarily unfavourable due to longer inundation (Vervuren et al., 2003). So, flooding allows propagule dispersion but also local extinctions. River corridor plants are very well susceptive to develop metapopulations in their river reach dispersion (Ouborg, 1993). For the conservation of these metapopulations, a sufficient number of dynamic populations is needed (Hegland et al., 2001). So, the observed isolation and fragmentation has a spatial but especially a temporal dimension, important for conclusions on conservation and restoration.

Restrictions in land use and management practices cannot stop gradual succession from pioneer to grassland communities, as these are governed by soil processes. So, the pioneer communities rich in river corridor species only survive under the benefit of flooding events with deposition of new sediments. Therefore, the rehabilitation of fluvial processes is a necessity. The rehabilitation of fluvial processes does not only mean that land use practices need to be changed to allow sediment zones to develop naturally, it also means that the river must transport enough coarse sediment. For this morphological criterion, sediment supply from eroding banks and gravel and sand bars in the river bed is a necessity. As these processes operate on a larger scale in time and space, a restoration approach at reach scale must be tailored to the shifting mosaics and patch dynamics of the dry river grassland habitat, with measures in the river bed as well as in the floodplain to assure the generation and rehabilitation of suitable habitat for the river corridor plants.
So, the identification of the recruitment limitation and the knowledge of habitat creation processes allows the design of measures in the river restoration programme. For effective conservation efforts for the endangered species, further knowledge of population biology and metapopulation dynamics are indispensable elements (Lowe, 2002), whereas the river corridor plants are a promising subject for metapopulation studies (Burkart, 2001; Menges, 1990). Research on several Meuse river corridor species for the metapopulation functioning and genetics has been initiated in 2003. First results (Jacquemyn et al., in Press) already confirm our observations of long distance dispersal and colonization with exceptional peak flow events for species with no adaptations to water propagation.

Conclusions

The river corridor plants are a good flagship species group for the protection and restoration efforts for larger Northwest European streams, as they cover a lot of information on the characteristic habitats and indicate well developed vegetations of the floodplain.
We detected the alterations to the floodplain dynamics as major threat for the river corridor plants in the present situation. Relicts isolated from the river flooding dynamics show no restoration potential, in contrast to relicts with high dynamics. The species that are cut off from flood contact by dikes, have the most serious isolation problem. Although we found the hypothesis on the conservation by preservation of relicts to be unsatisfactory, the protection of present relicts and newly generated habitat does need priority. The high recruitment potential of this endangered species group nevertheless tips the balance in favour of river dynamics restoration measures as most effective conservation approach. Habitat creation can be restored by changed river management and land use in the floodplain.

IV.3

Hydropeaking impact on a riparian ground beetle community

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Submitted to River Research and Applications

Abstract

The Common Meuse reach is strongly influenced by the operation of a hydropower plant at the upstream weir of Lixhe, especially during periods of low flow. Ecologically-based in-stream flow requirements and fluctuation thresholds have already been determined for this reach by reconstruction of the natural discharge course from historic and actual reference conditions. Nevertheless, more evidence from the present biota at risk has been demanded. This study therefore attempts to define boundary conditions for the low flow regime from the analysis of riparian ground beetles in this river reach. To achieve this, reference conditions for the hydroregime aspects of flow variation were determined. Then, using the habitat templet approach, the hydropeaking pressure was related to biological quality elements. Finally, after detecting the impact on the specific gravel bar ground beetles, thresholds and boundary conditions were determined for the hydropeaking pressure in peak velocity.

Introduction

River restoration projects generally aim to mitigate the effects of regulation works by rehabilitating geomorphological diversity, and promoting the recovery of degraded biota and the floodplain benefits from the river (Tockner & Schiemer, 1997). Problems can arise, however, when solutions are proposed without taking into account flow regime-related constraints (Kershner, 1997; Boulton et al., 2000; Gore et al., 2001). Economically as well as ecologically, low flows are a crucial element in the management of larger rivers of the temperate regions. Ecological criteria for low flow regime conditions are mostly addressed for in-stream flow requirements, in relation to deterioration of water quality and available habitat for lotic species (Gore et al., 2001). Impacts of hydropeaking were described mainly for aquatic species and communities (Scruton et al., 1997; Saltveit et al., 2001), yet no quantified rules have resulted from these studies. Constraints were also restricted to rapid flow decreases, as the emphasis was on stranding of fish and macroinvertebrates. Terrestrial riparian communities can be at risk as well, threatened by the rapid rise of water level. The responses of the riparian ground beetle community to hydropeaking pressures were therefore tested for the Common Meuse, where a reach-scale restoration programme is in development. This river restoration project implies measures of bed widening, bank lowering and flood channel restoration over a river stretch of 50 km (Pedroli et al., 2002).

Ecologically-based in-stream flow requirements and fluctuation thresholds have already been determined for this reach by reconstruction of the natural discharge course from historic and actual reference conditions (Salverda et al., 1998). Nevertheless, more evidence from the present biota at risk has been demanded. As riparian ground beetles have proven good indicators for riverbank habitat integrity and especially for flow regime conditions (Van Looy et al., 2005), we tried to define boundary conditions for the low flow regime.
The approach used follows general recommendations in defining boundary conditions for hydromorphological aspects in river restoration, as formulated in the guidance documents for the European Water Framework Directive (Wallin et al., 2003). Firstly, derivation of reference conditions for the hydroregime aspects of flow variation corresponding to no, or only minor anthropogenic alterations was made. Next, using the habitat templet approach (see Townsend et al., 1997; Van Looy et al., 2005), the hydropeaking pressure was related to biological quality elements. And finally, after detecting the impact, the responses of the specific ground beetle gravel bar templet were screened for thresholds that might reveal boundary conditions for this hydropeaking pressure.

Studied river stretch

The river Meuse has been highly regulated over the last 150 years, heavily influencing the flow regime, bed form and riverbank habitat conditions (Micha & Borlée, 1989). Hydroregime aspects of importance to the biotic system can be determined in baseflow conditions (Growns & Growns, 2001) and variability (Richter et al., 1996; Poff et al., 1997), as was documented for the Meuse by Van Looy et al. (2005). At the gauging stations of Stenay, Lorraine Meuse (France, see Figure 4.13) and Borgharen-Smeermaas, Common Meuse (Belgium), the Coefficient of flow Variation (CV) values over the last 10–100 years have been analysed by Jochems & Van Looy (2001). CV value ranges over 10 year summer periods for historical (1911–1920) and present day (1989–1998) data were calculated. The resulting values and their corresponding standard deviations are presented for Borgharen and

Stenay in Figure 4.14. The 1911–1919 CV values, representing Meuse discharges before large-scale flow regulation took place, are close to the Stenay values. The present day Borgharen CV values show a significant alteration in flow regime.

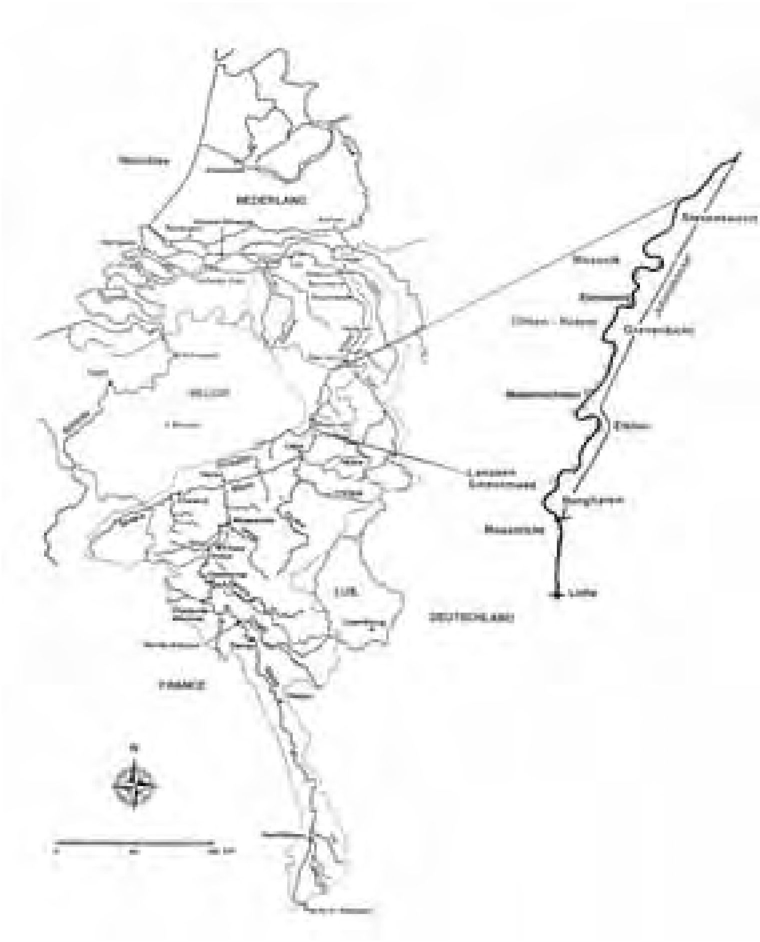


Figure 4.13 Map of the Meuse basin with inset for the reach level sampling stations and the position of the weirs of Lixhe and Borgharen.

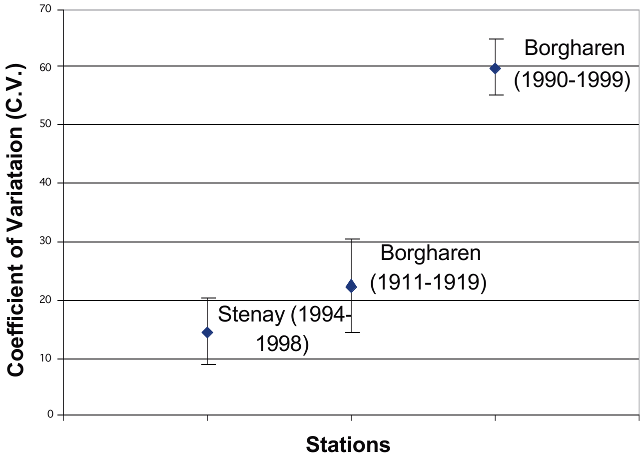


Figure 4.14 Comparison between summer mean Coefficient of flow Variation (CV) values (with SD) for upstream gauging station (Stenay), and Common Meuse present and historic situation (Borgharen present: 1990-1999 and historic:1911-1919).

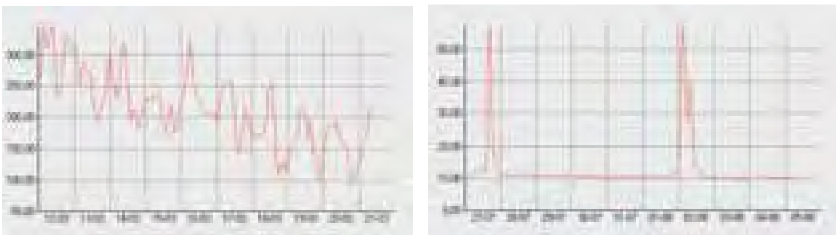


Figure 4.15 Hydropeaks (discharges in m^3/s) of the Common Meuse in spring (a) and summer (low flow)(b) at the gauging station of Smeermaas a few kilometres downstream the weir.

Summer discharge fluctuations are influenced significantly by weir management and water abstraction to canals and by the operation of a hydroelectric power plant at Lixhe, which, when functioning, is particularly influential on the hydroregime during low flow conditions (Figure 4.15). Due to water abstractions and weir management, the low flow conditions are extreme in terms of baseflow and duration, and under these conditions the plant releases of 80m?/s enter the Common Meuse as peak flows, with the water level rising more than one metre per hour.

The peak velocity – the increase in discharge within an hour, expressed as a percentage of the discharge at that moment – is very high at close proximity to the power station (41 at Smeermaas, the upstream gauging station for the Common Meuse reach), and reduces gradually over the 50 km reach to a value of 16 at Maaseik (the most downstream sampling station along the Common Meuse). The 80m?/s peaks flatten over the reach to increases of 20m^3/s.

Materials and methods

Sampling

In the summer of 1999 sampling was carried out on two gravel bank sites of the Common Meuse reach, 30 km apart (Maasmechelen and Elerweerd, see Figure 4.13). Thirty pitfalls were installed on each bar (six transects perpendicular to the river, with one pitfall in the steep bank zone and three on the gravel bar, making a grid over the site), and samples were taken daily for three weeks (30/6–8/7, 15–23/7 and 20–28/8). This fine-filtering sampling approach was executed in addition to the Meuse riverbanks sampling at catchment and reach level (See Van Looy et al., 2005).
The sampling at catchment scale was executed in 2000 using 14 stations spread along the middle to lower course of the river Meuse. The reach scale sampling of the Common Meuse was carried out for two consecutive years 1998 and 1999 on 17 gravel bank stations. Each station consisted of two plots; one higher on the riverbank and one close to the waterline, giving 34 plots in total. Ground beetles were sampled using pitfall traps (filled with 5% formaldehyde preservative), three traps in a row at 1m intervals forming a plot. Samples from the three traps were pooled and species identified in the laboratory. The traps were sampled every two weeks for the period May to October in both years.

Habitat templet approach at reach scale

Habitat templets of the River Meuse riparian ground beetles were derived from the catchment scale sampling, with a clustering and ordination of species, species traits and site conditions (Van Looy et al., 2005). Eight groups of ground beetle species were attributed to specific riparian habitats.
For this hydropeaking analysis we selected the habitat templet of the pioneer

gravel bars, the group of ground beetles living closest to the waterline. Significantly associated species traits in this templet are: wing development, dorsal flattening, small size and late season activity (Jochems & Van Looy, 2001). These predominant resilience/resistance traits have been documented in the literature for riparian habitat-dwelling Carabid species (Den Boer et al., 1979; Desender, 1989; Stork, 1990; Desender et al., 1994). The body size and phenology traits conform to those indicated for in-stream macroinvertebrates by Townsend & Hilldrew (1994). From the reach scale sampling and with input from the detailed site sampling, the habitat templet of the pioneer gravel bar has been refined in terms of species composition and species traits for use at reach scale. The resulting habitat templet description will be useful in the interpretation of the correspondence analysis, indicator choice and species response functions.
Once the expected species present within a habitat have been determined, the conditions that are favourable to the presence of these species can then be examined. So, detecting the direct impact of hydropeaking on this habitat templet species group is achieved by a diversity analysis over the reach (from highly impacted to non-impacted at the downstream end).

Analysis

Once an impact has been identified, the relationship between the pressure and the biotic indicator must then be analysed. For this purpose, we performed a correspondence analysis for the catchment data, followed by logistic regression for identified key predictor variables to screen for boundary conditions in the reach scale data.

A filtering of hydrological indices (see Van Looy et al., 2005) was performed for the hydropeaking effects on the habitat templet at risk, the pioneer gravel bar. Water rising speed and peak velocity were retained as hydrological indices for the correspondence analysis, and further environmental variables included were habitat heterogeneity and width-depth ratio of the sampling sites.

The catchment scale data taken from the pioneer gravel bar habitat templet (12 species) for 16 sample plots (with abundance > 80) were entered in a correspondence analysis. Based on the lower gradient length in the DCA, a redundancy analysis (RDA), carried out using CANOCO 4.0 (Ter Braak, 1988), was performed

to highlight interrelations between the environmental factors and species and to show the relevant environmental variables for this group.
The reach scale data taken from the pioneer gravel bar habitat templet for 62 sample plots (with abundance > 50) were entered in a correspondence analysis (Canonical Correspondence Analysis), followed by a multiple regression for the detected relevant variables. For the selected variables an ANOVA and multiple regression using STATISTICA (Statsoft Inc., 2001) showed the response of species diversity to these variables.

Results

The sampling of the two gravel bars yielded 6507 ground beetles from 71 species. In the day to day sampling, the ecological rationale beneath the habitat templets was revealed (Figure 4.16). With the waterline feeding strategy for the species group of the pioneer gravel bars being to forage on collembola and stranded organisms, the flow-related habitat condition of this community was highlighted, as already indicated by several authors (Hering & Plachter, 1997; Hering, 1998; Sadler et al., 2004). As these organisms are feeding immediately at the waterline, they are obviously sensitive to rapid rises of water level. The resilience traits predominating the templet of these highly disturbed sites (Townsend & Hilldrew, 1994) are key to their response to such a disturbance regime. Their ability to fly and swim allows them to endure a certain degree and frequency of habitat disturbance. This group of species and their predominant traits was defined for the pioneer gravel bar habitat templet of the Common Meuse and is shown in Table 4.9.

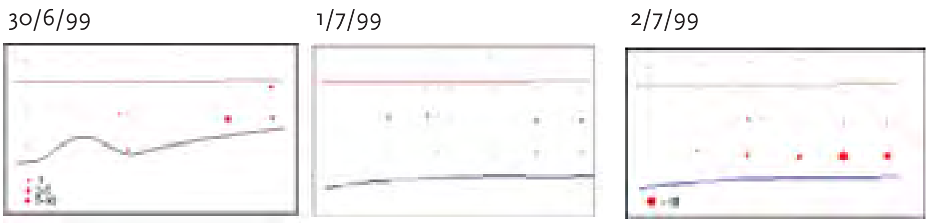


Figure 4.16 Indicator species of the pioneer gravel bar (Bembidion punctulatum) in the detailed riverbank survey. The blue lines represent the daily (mean) waterline, the size of the red dots indicates the individuals of the species sampled. The ground beetles were documented to follow the waterline in the day to day sampling after a small flow increment on 30/6.

Table 4.9 Habitat and life traits of the pioneer gravel bar habitat templet species of the Common Meuse.

Species	Vegetation	Substrate	Phenology	Dispersion	Ecological group
Amara aenea	Scarce vegetation	No preference	Late Spring	High, macropteric	Stenotope, xerofilic
Anisodactylus binotatus	High vegetation of grasses, sedges	Open, wet clay and sand	Spring macropteric	High, stenotope, hygrofilic	Modestly
Bembidion decorum	Unvegetated riverbanks	Gravel, sand	Late Spring	High, macropteric	Stenotope, hygrofilic
Bembidion punctulatum	Unvegetated riverbanks	Gravel	Late Spring	High, macropteric	Stenotope, hygrofilic
Bembidion atrocoeruleum	Unvegetated riverbanks	Gravel	Late Spring	High, macropteric	Stenotope, hygrofilic
Bembidion testaceum	Unvegetated riverbanks	Dry gravel, sand	Spring	Low, brachypteric	Modestly stenotope, xerofilic
Bembidion lampros	Open terrain	No preference	Spring	High, wing-dimorphism	Very eurytope
Harpalus affinis	Open vegetation	Gravel, sand, loam	Spring and autumn	High, macropteric	Modestly stenotope, xerofilic
Panagaeus bipustulatus	Open medium-wet grassland	Dry gravel, sand	Spring	High, macropteric	Modestly stenotope, xerofilic
Perileptus areolatus	Unvegetated riverbanks	Gravel or coarse sand	Late Spring	High, macropteric	Stenotope, hygrofilic
Thalassophilus longicornis	No preference	Very wet gravel and coarse sand	Spring macropteric	High, hygrofilic	Stenotope,
Trechus quadristriatum	Mosaic vegetation	No preference	Autumn	High, macropteric	Stenotope, hygrofilic

Over 16 000 carabid beetles were examined and identified from the reach level sampling carried out in 1998 and 1999. The catchment sampling yielded 4892 ground beetles extracted from the pitfalls and were determined to species level. Redundancy analysis for the environmental variables and the pioneer gravel bar community (12 species) at catchment scale showed peak velocity to be the environmental variable with the highest biplot score (Figure 4.17), as did the CCA at reach level (see Van Looy et al., 2005). Figure 4.17 shows a group responding to the water rising speed and peak velocity: Harpalus affinis, Bembidion testaceum, B. decorum, B. punctulatum and Panagaeus bipustulatus. Their predominating traits are small body size and no or minimum developed wings, making them vulnerable to the rapid flow increases. Still, they are quick colonizers of the open riparian

habitat, so their presence on the gravel bars is unaffected by the habitat aspects of surface and higher refuge. A second group of species – responding to the first axis, with Bembidion lampros and Amara aenea – is related to habitat heterogeneity and width–depth ratio, and shows a less strict habitat preference and are not strict xerophilic species. This group selects the well-established larger gravel bars, offering enough refuge for peak flows.
The impact of hydropeaking on the species group of pioneer gravel bars is indicated by the increasing average richness along the reach (Figure 4.18). Peak velocity is the environmental variable best representing the hydropeaking effect, showing a similar linear trend over the reach – albeit opposite to the species richness.

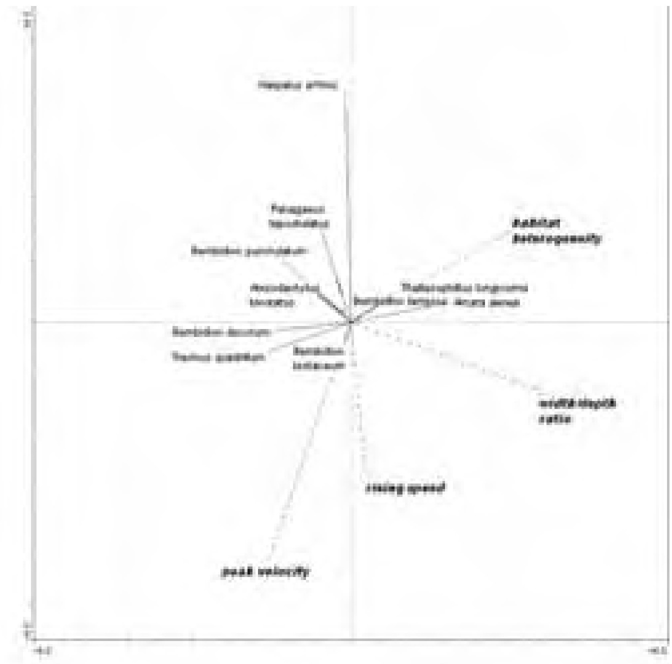


Figure 4.17 Redundancy Analysis biplot of the sampled pioneer gravel bar habitat templet and the environmental variables.

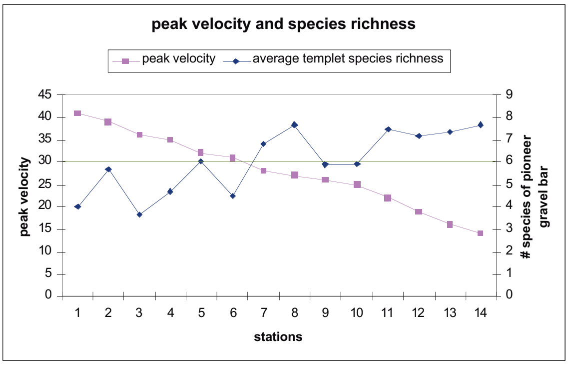


Figure 4.18 Average species richness for the pioneer gravel bar templet over the plots of the Common Meuse stations (green line indicates optimising peak velocity value).

ANOVA revealed a significant relationship between species diversity in the pioneer gravel bar habitat templet and peak velocity over the Common Meuse sampling plots (F:315.12, $p < 0.0001$) (Figure 4.19). With multiple regression, a significant regression function was derived for the species diversity (beta=-0.56, F:29.9, $p < 0.0001$).

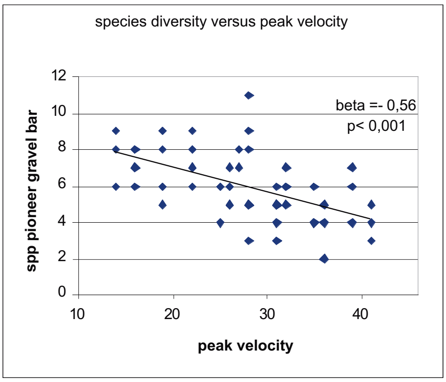
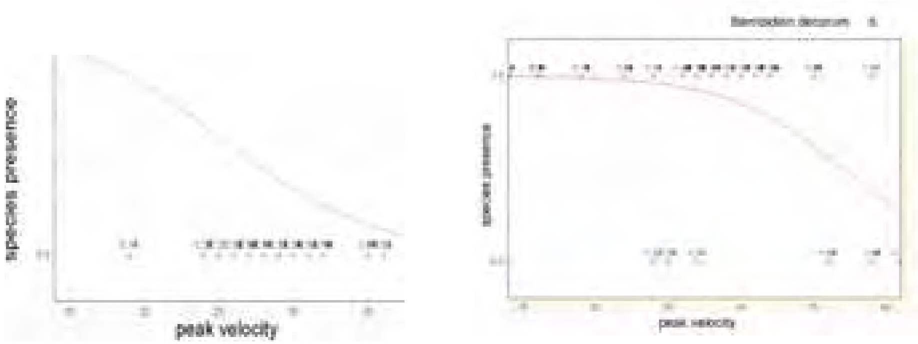


Figure 4.19 Species diversity of the pioneer gravel bar templet group(14spp) for the Common Meuse plots, related to peak velocity with the regression function.

The linear regression for species diversity shows the optimum conditions for carabid communities of the dynamic habitats in the zones where the human-induced discharge fluctuations are dampened; a point also illustrated by the average plot species richness in Figure 4.18. The responses of indicator species to peak velocity using logistic regression (Harpalus affinis and Bembidion decorum, respectively chi^2= 25.9, p<0.001 and chi^2= 22.1, p<0.001) confirmed significantly the threshold value of peak velocity as 30 (Figure 4.20).



a. b.

Figure 4.20 Logistic regression results of the indicator species for the peak velocity, Harpalus affinis (fig a), and Bembidion decorum (fig b) showing the threshold peak velocity value of 30 (30% discharge increase per hour) as the bending point in the logit presence-absence regression function.

Discussion

Research into low flow regime conditions is an expanding field in light of integrated water management and sustainable water use being confronted with water shortages and strong regulation impacts.
In-stream flow evaluations are mostly based on single-species approaches, or combinations of target (mainly fish) species' habitat availability (IFIM, PHABSIM, Bovee, 1985; Stalnaker et al., 1995). Gore et al. (2001) reviewed macroinvertebrate in-stream flow habitat requirements, useful in stream management and restoration. They concluded that including benthic macroinvertebrate diversity in fish-based evaluations showed significant differences, especially for minimum flow requirements. Growns & Growns (2001) demonstrated the impact of flow regulation on aquatic macroinvertebrate and periphytic diatom communities. Their results showed significant effects of hydropeaking and indicated different responses for different habitats studied in the impacted rivers.

Ward & Stanford (1979) illustrated potential effects of different kinds of flow regime modifications on zoobenthos, with emphasis on the factors controlling available habitat and drift. They stressed hydromorphological effects of flow modifications on availability of food and substrate for this aquatic community. These effects on current velocity, depth fluctuations and turbidity correspond to those indicated for other groups; the bed and bank instability are specific to this group. This impact of flow regulation on bed and bank structure was also documented for in-stream habitat conditions (Walker et al., 1979), as well as for riparian vegetation (Kauffman et al., 1997; Sparks et al., 1998; Friedman & Auble, 1999). But this latter effect results more from general flow regime alterations such as duration and level of low flows, and less from hydropeaking.

Hydropeaking studies are mainly focussed on the falling limb of the peak hydrograph (rapid flow decreases caused by hydropeaking), with effects of changes in current speed or dessication, causing drift or stranding of organisms (Cushman, 1985; Valentin et al., 1994; Saltveit et al., 2001).
Our study is novel in this respect because it emphasizes the biotic responses and impact of hydropeaking on the peak's rising limb. We identified this response in a significant relationship between the habitat templet group species richness and peak velocity.

The habitat templet theory has been applied for hypothesis testing of species responses to disturbance (Townsend & Hilldrew, 1994; Townsend et al., 1997). Here, we explore the use of the habitat templet approach to derive boundaries for specific hydroregime conditions, and the research outline gave strong confidence to the detected responses. As we started from a multiscale, over-year observation of communities and species, the habitat templet approach in combining species traits to grouping and deriving relationships to the physical environment, proved useful for our purposes. The observed spatial and temporal patterns in species distribution over the riparian zone, detected in the local detailed study, were useful to interpret the overall observed species assemblages and trends. Indicative power for the correlation results lie in the sampled abundance of the indicator species (Bembidion decorum n= 1968 and Harpalus affinis n= 201) and the fact that this is their preferred habitat and they have well established populations over this river reach.

As we detected a significant impact and pressure response, we were able to identify boundary conditions, thanks to the gradual dampening of the pressure over the river reach and the multiscale approach of our investigation. The presence of unimpaired sites – in the upstream reach as well as in the most downstream sampling stations for the Common Meuse reach – and the screening for a range of environmental variables over the locations, allowed the identification of peak velocity as a critical factor, plus the detection of a threshold value because the whole pressure gradient was sampled over the Common Meuse reach. This threshold value can be proposed as a boundary condition for large gravel-bed rivers with hydropeaking problems. For smaller highland rivers, thresholds for the communities of that river type might be higher, whereas in lowland rivers, lower boundary conditions can be expected.

Our analysis adds strongly to the method of natural flow reconstruction (Poff et al., 1997; Salverda et al., 1998), as it gives a tangible measure for critical boundaries and the pressure–impact relationship. Our results gave a comparable measure for the critical peak velocity as obtained with the flow reconstruction method (Salverda et al., 1998).
Possible sustainable Common Meuse recolonisation of species of this habitat present in the upstream part of the catchment (e.g. Thalassophilus longicornis, Perileptus areolatus, Bembidion elongatum) will depend on the habitat quality (influenced by hydropeaking pressure) and the great distances. Argument for protection of riparian ground beetle fauna on a larger spatial scale, in view of the evidence of habitat fragmentation and dispersal limitations, has already been provided by Andersen & Hanssen (2005).

Conclusion

At the weir of Borgharen, measures were taken to dampen the strong fluctuations caused by the turbine releases of Lixhe, and so for the optimisation of the sluices and weir management criteria for an acceptable fluctuation, further research was needed. From our analysis, boundary setting was possible for this specific hydromorphological pressure. The results indicate that fluctuations of the Common Meuse low flow regime should be dampened by ? to reach an acceptable peak velocity value of 30. Amelioration of the situation can also be aided by the proposed restoration measures. Widening of the riverbed is a very successful measure in dampening discharge fluctuations. Hydropeaks of 80 m^3/s entering the Common Meuse can be topped by the enlargement of the river bed. The bed enlargement of the first restoration site of the reach can be designed in such dimensions that it dampens the hydropeaking impact to an acceptable peak velocity. In conclusion, we can state that this habitat templet analysis revealed tangible measures for the hydraulic management and the rehabilitation project.

V Tools for Assessment and Evaluation

For the defined key ecological factors and steering processes (Chpt III) and their identified response rules and thresholds (Chpt IV) a set of tools for assessment and evaluation can be generated, useful in the development of restoration programmes, in the design and choices in conservation and restoration and in the definition of conservation objectives. First we refer to the reference conditions and the use of references in this process.
Secondly, we integrated the determined relationships between the key ecological factors into a dynamic model approach, useful in the prospection of restoration potentials and scenario's.
Thirdly, we developed an evaluation method, as a tool for spatial planning in the light of combining sustainable flood protection and floodplain rehabilitation/river restoration at river basin scale.

All relevant scales in integrated river management: ecoregion, catchment, reach and local scale, are emphasized in this chapter. Starting from a set of comparable rivers at ecoregion level, to the River Meuse over the three relevant river scales.

For the processes, especially the interaction between the physical and biotic processes is under study in this chapter. Major group is the hydromorphological processes, for which first reference conditions are derived, an further the responses of the biotic system are integrated in the model approach, to end up with the evaluation over the different gradients in the river system.

Central questions are for the reference conditions, useful in defining the restoration potentials. The modelling approach is integrated to answer the question to where restoration can lead to and how we can prospect this restoration endpoint, immediately raising the final question for evaluation.

Themes and groups emphasized upon are riparian forests for the reference conditions at ecoregion scale, the riparian ground beetles and floodplain meadows at river scale in the evaluation and all groups for the reach scale modelling approach.

V.1

Including riparian vegetation in the definition of morphological reference conditions for large rivers; a case study for the European Western Plains.

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Submitted to Environmental Management

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Abstract

Methods for defining and retrieving reference conditions for large rivers were explored with emphasis on hydromorphological and biological quality indicators.

Boundary conditions for riparian zone functioning were investigated for hydromorphological and riparian forests characteristics. Critical ranges for riparian forest area, for stages of riparian forest development and for sustainable populations of Populus nigra and Salix purpurea were determined in the search for useful measures from reference conditions.

After identification of reference conditions, a proposal for assessment and monitoring of the proposed indices is discussed for its applicability.

Introduction

Human impacts alter the natural hydromorphological conditions in rivers. As hydro-morphology determines for a large part the ecological conditions, these pressures have severely altered the ecological status of rivers. The European Water Framework Directive (WFD, EC, 2000) demands a quantitative assessment of the ecological status and obliges the member States to achieve a "good ecological status" (or good ecological potential) of all waterbodies by 2015. The ecological status refers to "the quality of the structure and functioning of aquatic ecosystems" (WFD, art. 2). A good ecological status (or potential), is a "slight deviation" from type-specific reference conditions (or maximum potential). These reference conditions have to be derived for all types of water bodies and represent the values of the biological quality elements at "high status", i.e. where "there are no, or only very minor, anthropogenic alterations of the physico-chemical and hydromorphological characteristics" (WFD, annex V, table 1.2). Reference conditions may be based either on historical or geographical comparisons or on modelling, or may be derived using a combination of these methods including historical data. Especially for larger rivers, defining reference conditions however, proves to be problematic. Historical data on structure and functioning of the freshwater ecosystem prior to degradation are often missing, and there are few, if any, modelling methods directly applicable to large rivers. This problem can be overcome through the identification of appropriate current reference sites as the guide. In selecting analogue sites, the typology of the rivers must be carefully evaluated. Inherent differences among locations in geology, climate, position in the catchment, fluvial geomorphology, hydrology and biogeography must be considered (Pedroli et al. 2002, Palmer et al 2005).

Our research was tailored towards the definition of morphological reference conditions in western European large rivers. Among the hydromorphological parameters required by the WFD for the classification of ecological status, alongside of hydrological regime and river continuity, the “structure of the riparian zone” and the “river depth and width variation” are explicitly mentioned. The width-depth ratio is a good abiotic indicator of morphological alterations (Raudkivi, 1998), but the relationships between the magnitude of the changes in river geometry and the related biological impacts are poorly documented. On the other hand, riparian vegetation and especially riparian forests are yet documented as good integrative indicators for the hydromorphological conditions of large rivers, as was acknowledged in several international research programmes (Lefèvre et al. 2001, Hughes 2003). Riparian vegetation structure and dynamics not only respond to direct morphological alteration such as river channelization, but also to hydrological changes, especially the flood regime, and to the connectivity to alluvial aquifers (Décamps et al. 1988, Bornette & Amoros 1996, Girel & Manneville 1998, Grevilliot et al. 1999, Van Looy et al. 2003, Naiman et al. 2005), both parameters also required for the evaluation of hydromorphological conditions.

Starting point for this collaboration was the difficulty encountered in the definition of reference conditions for the heavily modified river Meuse. Human activities in the Meuse catchment’s area causing alterations in the hydromorphological conditions of the river system run back to the earliest cultivation of land. The river Meuse was also the artery of Europe’s mainland first industrial revolution. A chain of industrial and human settlements (larger towns) borders the river and the use of the river as major waterway goes back in time even much further to Roman times. As the historic reference condition of the river Meuse is hard to define and especially quantitative data mostly lacks for historic situations (Micha & Borlee 1989), references have to be searched elsewhere. For this purpose all large rivers of the European Western Plains ecoregion (as defined by the WFD) were screened for the presence of actual references for the river Meuse. The four large rivers of this

ecoregion draining the adjacent Western Highlands ecoregion, Meuse, Loire, Allier and Dordogne show comparable gravel reaches just downstream their Western Highlands stretches (Figure 5.1). They are rain-fed rivers with no annual snow melt discharge peaks in spring but exceptional peak flows with short duration (flash flows) in periods with high precipitation.
Through the case study dealing with these four western European large rivers, we tried to address the following issues : 1) how to select reference sites in large rivers on the basis of hydromorphology, 2) what riparian vegetation measures can be included in the reference conditions and more widely in the ecological status evaluation, and 3) how these measures relate to morphological alterations, evaluated especially by the width-depth ratio. Finally, by evaluating the minimum requirements for a sustainable riparian functioning, indicative values are proposed for the setting of “good ecological status” boundary conditions in large rivers.

Defining criteria for Hydromorphological Reference Conditions in large rivers

According to the WFD (annex V), reference conditions shall represent the values of the biological quality elements at “high status”, i.e. where “there are no, or only very minor, anthropogenic alterations to the values of the physico-chemical and hydromorphological quality elements”. Reference conditions may be based on existing reference sites, or on modelling, or can be derived using a combination of these methods including historical data.

The REFCOND guidance (Wallin et al. 2003), resulting from a wide discussion among European experts, gives a consensual interpretation of the reference concept:

- “Reference conditions (RC) do not equate necessarily to totally undisturbed, pristine conditions. They include very minor disturbance which means that human pressure is allowed as long as there are no or only very minor ecological effects;
- RC equal high ecological status, i.e. no or only very minor evidence of disturbance for each of the general physico-chemical, hydromorphological and biological quality elements;

- RC can be a state in the present or in the past."

This interpretation offers more realistic possibilities for defining a reference state, as it refers to the effective ecological impact of physical alterations; in this sense, both have to be evaluated simultaneously to assess a "very minor" ecological effect, i.e. hardly distinguishable from the natural variability of the system.

However, for rivers morphodynamics, it is difficult to refer to a particular "state". Rivers are highly dynamic systems, primarily controlled by physical factors. Three key words could define the river ecosystem functioning: processes, dynamics, and reversibility. In a "healthy" river, functional processes (hydromorphological, biogeochemical, ecological) maintain a physical and biological dynamic state, which ensures the reversibility of the system after natural and anthropogenic disturbances. Stable systems are either a typological exception (like lake outlets), or the result of an anthropogenic regulation (Roche et al. 2005). In this sense, hydromorphological reference situations can be searched in reaches where the fundamental physical processes are not altered. Reference conditions can then be defined as the range of variability (spatial and temporal) of the observed physical and ecological structures. These RC are not fixed over time. Long term (> 100 years) morphological evolutions are observed due to natural climatic changes, and to land cover alterations. But if the connectivity is maintained, and the biodiversity conserved at the basin scale, the biocenosis will adapt itself permanently to a physical system in dynamic equilibrium (Roche et al. 2005).
To focus on the current possible causes of alteration of the fundamental processes, rather than to a past "state", will help to define RC.

A "pristine" state, without any impact of human activities in the river basin, offers no perspectives in the definition of reference conditions for our western European rivers. However, we can define a "natural" state if we accept the assumption that, due to the intrinsic resilience of river systems, man can live in a basin without significantly degrading the river's ecological functioning and biodiversity. Such a natural state will correspond to a "free" river, with very limited impact of artificial structures. Hydro-sedimentological, biogeochemical, ecological processes are still active, and only slightly altered by land use in the catchment; the morphodynamic processes and the connectivity with the floodplain are maintained, although sometimes spatially limited (Wasson, 1992). For many modified rivers, this natural state refers to conditions before the large regulation activities of the 19th century.

We propose that criteria for reference thresholds of hydromorphological conditions in large rivers correspond to the beginning of the fundamental structures and processes alteration. This could be evaluated by looking both at the pressures acting at the basin and reach scales, and at the morphological responses at the reach scale. Criteria for the pressures could be:

- (1) at the basin scale, the regime and fluxes of water and sediments are not significantly altered by impoundments or land use, as compared with a natural vegetation cover.
- (2) at the reach scale, the fundamental morphodynamic processes are not significantly altered by artificial lateral constraints, and the river has the necessary "space of freedom" to maintain the possibility of a dynamic adjustment.

For the morphological features at the reach scale, the following criteria have to be met:

- (3) the morphological type correspond to what could be expected as result of a dynamic equilibrium in the present climatic conditions, owing that the condition (1) is fulfilled;
- (4) all the important side channel structures are still significantly present;
- (5) Lateral connectivity is maintained at the reach scale;
- (6) Type specific riparian ecosystems are still present in significant areas.

In search of a hydromorphological reference for the river Meuse.

Studied sites

In the Meuse catchment's area, human activities causing hydromorphological alterations run back to the earliest land cultivation, and the use of the river as major waterway goes back to Roman times. In more recent times, the River Meuse was the artery of Europe's mainland first industrial revolution. A chain of industrial and human settlements including large towns borders the river. For the heavily altered studied reach of the Meuse river, historical quantitative data are lacking, and reference conditions have to be searched elsewhere. For this purpose, all large rivers of the European Western Plains (ecoregion 13 as defined by the WFD) were screened for the presence of actual reference sites for the River Meuse. Three river reaches in the ecoregion 13, draining like the Meuse the adjacent Western Highlands (ecoregion 8), were selected as possible reference situations as they exhibit more natural morphological features; these reaches belong to the rivers Loire, Allier and Dordogne (Figure 5.1).

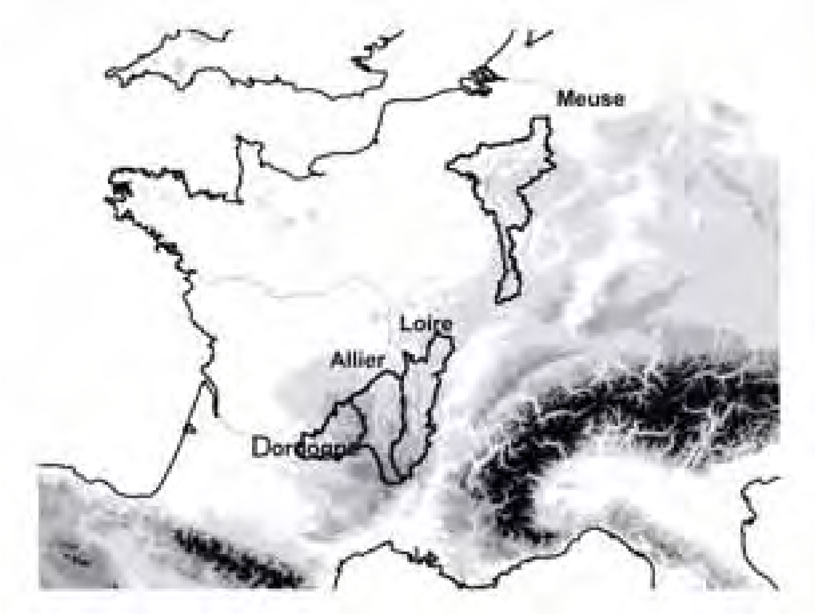


Figure 5.1 Location of the studied river reaches in North-West Europe with the upstream catchments delineated on an altitude background.

The studied reaches of the four rivers were selected on the basis of the general morphological character of valley form, slope, discharge and sinuosity. The chosen stretches are located around kilometre 300-400 of the rivers. For the Meuse with its narrow upstream basin and large subcatchment of the Ardennes Massive drained by the middle part of the river, the studied downstream reach is around km 450 just downstream Maastricht. For the Dordogne coming from the 'Parc des volcans d'Auvergne' in the Massif Central, the stretch Souillac-Vitrac just before kilometre 300 was chosen. For the Allier and Loire flowing northwards from the Massif Central, reaches around km 370 were chosen, between Châtel-de-Neuvre and Moulins for the Allier, and around Lamenay for the Loire.

Comparison of the sites typology

In order to define hydromorphological reference conditions, we first had to ensure that these four reaches could be classified in the same morphological river type. For this purpose, we examined the geophysical and climatic characteristics of the basins, and the size, hydrological regime and morphological characteristics of the studied reaches.

The basin's characteristics are summarized in table 5.1. The altitude range is higher in the Loire, Allier and Dordogne rivers (300 – 1700m), flowing out from the French Massif Central, than in the Meuse (100 – 700m) coming from calcareous hills and then crossing the Ardennes massif. In the framework of the EU funded REBECCA project, an ongoing work for the definition of hydro-ecoregions allowed the characterisation of litho-morphological structures at the European scale; from these data we evaluated the percentage of each basin that could be classified as middle mountains, hills and plains, with the corresponding lithological features. All four basins are dominated by crystalline (granitic and metamorphic) rocks, but the proportion is lower in the Meuse basin (55%) than in the three others (> 80%). The rivers coming from the Massif Central have a large proportion (58% to 72%) of their basin classified as middle mountains, while the Meuse basin correspond to hilly landscapes (88%). The Meuse, Loire and Allier reaches are situated in clayed or phreatic alluvial plains, while the Dordogne alluvial valley entrenches a calcareous plateau. The climatic conditions are comparable in the four basins, with 800 to 1000 mm of annual rainfall.

Table 5.1 Upstream basin characteristics (altitude, lithology, precipitation from REBECCA data).

Altitude (m)	**Meuse**	**Loire**	**Allier**	**Dordogne**
min	20	190	215	89
max	687	1.631	1.726	1.756
mean	**273**	**568**	**711**	**666**
std	117	291	308	290
Litho-morphological regions	**Meuse**	**Loire**	**Allier**	**Dordogne**
Crystalline middle mountains		58%	65%	72%
Crystalline hills	55%	24%	19%	14%
Calcareous hills	33%	2%		14%
Calcareous tabular plains	4%			
Clayed plains	5%			
Phreatic alluvial plains	3%	16%	16%	
Precipitation (mm)	**Meuse**	**Loire**	**Allier**	**Dordogne**
min	744	715	701	893
max	1.201	906	1.058	1.049
mean	**992**	**797**	**820**	**954**
std	108	44	78	32
sources altitude: MNT KM USGS,	precipitation: Climatic Research Unit, university of East Anglia			

At the reach scale (Table 5.2), upstream catchment was calculated from the upstream point of the selected reach. The catchment areas vary from 8700 km^2 for the Dordogne to 20200 km^2 for the Meuse. The hydrological regimes are similar (Figure 5.2), all four rivers are rain-fed without high snow melt discharge peaks in spring, but exceptional short duration peak flows are possible in high precipitation periods. Due to a larger basin area, winter discharges are higher in the Meuse. Floodplain width is the average of the natural floodplain area; for the Meuse the disconnection by winter dikes isolates large parts of this floodplain. The floodplain natural width, the wavelength and sinuosity lie in the same range.

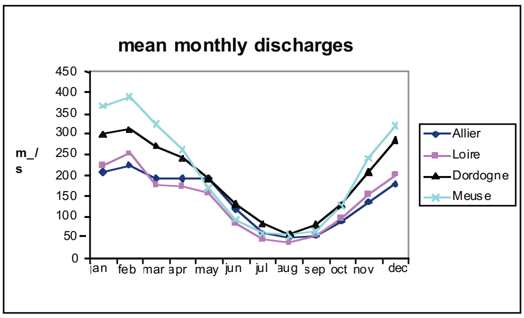


Figure 5.2 Hydrological regime at the reach scale.

Table 5.2 Hydrological and geomorphological characteristics of the selected reaches.

	Allier	Loire	Dordogne	Meuse
Length, distance from source (km)	336-368	370-402	280-306	450-485
Upstream catchment area (km^2)	12400	14200	8700	20200
Floodplain width (m)	1400	1100	950	1250
Wavelength (m)	1850	2050	2100	1900
Sinuosity	1.39	1.24	1.27	1.35
Max. measured discharge (m^3/s)	1200	2080	2300	3200
Mean annual maximum discharge (m^3/s)	710	980	1100	1250
Mean annual minimum discharge (m^3/s)	21	18	21	10
Mean annual discharge (m^3/s)	120	140	170	170
Slope, elevation difference between stations at the start and end of the reach divided by river length	0.6	0.68	0.73	0.49
Width, average values (m) for the studied reaches over the bankfull sections	150	120	110	100
Depth, average values (m) for the studied reaches over the bankfull sections	4	5	5	7
Width-depth ratio, W/D	38	24	21.5	14.5
Bankfull discharge, Qbf (m^3/s)	850	1100	1200	1600
Stream power, $\Omega=\gamma.Qbf.S$ (W/m) with $\gamma=\rho.g=6.25$, Qbf= bankfull discharge and S= slope	3188	4675	5475	4900
Natural stream power, $\Omega=\gamma.Qan.S$ (W/m) with Qan the mean annual maximum discharge	2663	4165	5019	3828
Specific streampower, Ω/width	21	37	50	49
Natural specific stream power, Ωn/width	18	35	46	38
Bed texture D50 values in mm	4	7	12	35
Bar texture D50 values in mm	1 - 5	1 - 5	5 - 10	1 - 25
Bank texture (in mm, 25-75percentiles)	0.5-1.4	0.3-1.1	0.3-0.9	0.1-1
Embankment (% of linear length)	2	11	14	90

Gauging stations Allier (Châtel-de-Neuvre) 1986-2002, Loire (Gilly-sur-Loire)1955-2002, Dordogne (Souillac) 1973-2001, Meuse (Smeermaas) 1978-2002, (sources: ministère de l'écologie et du developpement durable, données bassins ; Meuse, DIHO)

For the four reaches, river hydromorphological parameters are derived starting from the general bed geometry with the frequently used width-depth ratio, then integrating hydrometrics with the measure of bankfull discharge, and the derived 'streampower' measure, integrating the slope of the reach, and with a annual discharge measure a natural streampower measure, as the bankfull discharge is severely altered by alterations. Even more geometry is integrated with the 'specific streampower', the streampower per unit of bed width. The measure of bankfull width varies strongly for meandering reaches. For the four stretches, the values ranged between: Allier 120-200, Loire 110-150, Dordogne 90-125, Meuse 80-120. Local hydromorphological conditions are entered with the bed substrate characteristics of bed and bar texture. Sediment substrate of bars shows D50-ranges from 1-25mm for the surveyed Meuse bars, for the Allier and Loire values are between 1-5mm (D50), along the Dordogne there are also coarser sediment bars present (D50: 5-10mm). Especially the Allier has a large fraction of sandy bed load, which is not washed out as quickly as in the narrowed channels of the other studied river stretches. Added to these river parameters, the human modifications in the form of embankments were recorded. The Allier reach is a nature reserve area (Reserve Val d'Allier) and bank protection is only present near civil works of bridges. For the Loire and the Dordogne, most meander bends are consolidated and local encroachments are present to protect infrastructure like campings. The Meuse is a highly normalised stretch.

The specific streampower is very high in the incised river reaches of the Meuse and the Dordogne. For the Dordogne, this high value is due to the stronger slope of the stretch. As the bankfull discharge is affected by this incision process, a 'natural streampower' measure was calculated with the mean annual discharge value. The bankfull discharge (Qbf) is estimated from field survey results and cross-sections of the river. Starting from the 2-year recurrent discharge peak, for the Allier but also for the Loire these values were lowered to the 1,5 year recurrent flows, as there is a frequent contact between the river and floodplain, and bed incision is not clearly present in the cross-sections. For the Meuse, the bankfull discharge is calibrated with the measurements over the reach. The strong bed incision (average 3-4 meter over the reach for the last 100 years) results in bankfull discharges corresponding to a less recurrent discharge (once in 3 years). For the Dordogne the river bed lies also deep in its valley floor and the 2-year peak discharge was retained as Qbf.

So, natural streampower for the Meuse is lower than for the Loire, yet, actual streampower is higher. This can be observed in the high scouring for the Meuse river bed, with only coarse gravel remaining and forming an armoured layer (D50: 35mm).

On the basis of their basin characteristics, we can infer that the rivers flowing out from the Massif Central (Allier, Loire, Dordogne) pertain to the same type, but the litho-morphology of the Meuse basin is slightly different, with a lower altitude and less crystalline rocks, which could influence the river morpho-dynamics. All four reaches have similar size, hydrological regime and floodplain width, but we have to verify that their natural morphological type would be the same. This can be first inferred from the sinuosity (close to natural values for the Meuse main channel), which lies in the same range for the four reaches. From the comparison of the natural stream power and the bank's granulometry, where sand predominates (Table 5.2), we can also expect a similar potential river bed dynamic. However, the best evidence is given by historical maps (around 1800) of the Meuse and Loire reaches (Figure 5.3): both rivers had the same morphological features before large regulation activities took place. We can thus consider that the four reaches belong to the same morphological type.

Figure 5.3 Historical maps (1800) of the Meuse (left) and the Loire (right) stretches (upper 10km of the selected reaches) show the resembling hydromorphic conditions before the larger regulation activities took place.

Identification of a potential reference reach

In this phase we have to look for the hydromorphological pressures and impacts, and screen for the criteria we've listed under par. 2. In table 5.3 the relation between the main hydromorphological pressures and impacts for the selected rivers, and the in this paper discussed measures for the hydromorphological conditions, are presented.

Table 5.3 Hydromorphological pressures and impacts emphasized in the paper.

	object	measure	specification
hydromorphological pressure	dams, weirs	flow variability, sediment granulometry	upstream pressure with possible downstream hydrological and morphological impact
	embankment	embankment%	local pressure with more or less local impact
hydromorphological impact	bankfull width	Qbf	encroachment of the channel
	depth	width-depth ratio	bed incision
landscape impact	riparian dynamics	riparian dynamics indicator	erosion-sedimentation zones
biological impact	riparian forest communities	riparian forest extent	area/river kilometre
	riparian forest recruitment	young forest stages	area/river kilometre
	target species	target species populations	# populations/river kilometre

The flow variability can be measured in a periodical variation of flow with a Coefficient of flow Variation, Qt/Qt-1, as defined by Growns & Growns (2001), or in a mirror of amplitude ranges as in the Range of Variability Approach (Richter et al.1996). Sediment granulometry is important in view of the meandering processes for the erodibility of riverbanks, emphasized to the bank granulometry, or can be regarded with respect to actual sedimentation and morphological character of the reach.

Impacts to bankfull dimensions are already described as caused by normalisations.
The riparian landscape dynamics for the four reaches, were related to the hydromorphological pressures measured in the embankment and the hydromorphological impacts in the most common used river parameters in the description of hydromorphological character: width-depth ratio, bankfull discharge (Qbf), streampower and specific streampower.

We can verify whether the Allier can be retained as reference reach, based on the criteria we've listed under par. 2. Although there are some weirs and impoundments in upstream sections, they do not significantly affect the flow regime or sedimentological character of the river at the studied reach. At this reach scale, morphodynamic processes are more or less natural for this meandering reach, there is no sign of normalisation or embankment impacts, no artificial levees are present and there is a large natural riparian corridor present. These observations can be supported with an aerial photograph of the studied Allier reach, contrasting to the heavily modified situation of the Common Meuse reach (Figure 5.4). The Allier can thus be proposed as potential reference reach.

Figure 5.4 Aerial photographs of the Allier and the Meuse study areas in the present situation (photographs 2003).

Defining reference conditions and selecting measures for the biotic integrity of the river corridor's hydromorphological character

Survey of riparian forest communities for the four reaches

Alluvial forests have virtually disappeared from most of the large river valleys of North-western Europe, making way for cultivated land and meadows. Large gravel bed rivers are edged by riparian formations of willow and poplar (Salicion albae), further in the floodplain replaced by elm and ash alluvial forests (Ulmo-Fraxinetum) and higher elevated grounds gradually develop oak forests (Querceto-Ulmetum). The riparian forests of Salix and Populus are designated priority habitat in Europe's Natura2000 conservation strategy (Habitats Directive, EC, 1992).

We distinguished 5 communities within the riparian softwood forests of the Salicion albae for these large gravel rivers (nomenclature follows Schnitzler 1997):

- Salix purpurea thicket or young Populus nigra formations.
- Willow thickets dominated by Salix alba.
- Salici-populetum forests.
- Salicetum albo-fragilis.
- Dry populetums

Characteristic species in these communities are Populus nigra and Salix purpurea. These species play a key role in the morphological development of the riparian zone of large gravel rivers. The accretion of bars and islands depends on these species for their capacity to catch sand and hold the developing sediment zones (Hughes et al. 2001, Van Looy et al. 2005a). With their highlighted problems of gene flow and recruitment limitation (Imbert & Lefèvre 2003, Vanden Broek et al 2004), emphasis on these species and their populations and habitat potential is important in the scope of this study.
Stand conditions, population dynamics and genetics of these species are subject of many research programmes and networks have been installed to develop conservation strategies for these species and the ecosystems they belong to (Lefèvre et al. 2001, Guilloy-Froget et al. 2002). Successful conservation strategies for these species need to consider the current status of existing populations as well as the physical dynamics of the natural habitat formed by the river. For Black poplar a European conservation programme for this species has been installed and for the Meuse a reintroduction programme initiated (Vanden Broeck et al. 2004)
In the riparian forests of the Loire and Dordogne the presence of the exotic species Box alder (Acer negundo L.) is noteworthy. It is present in the riparian forests of the Loire and most common along the Dordogne, as it is remarkably widespread present along the rivers of the western part of this ecoregion. It is mostly restricted to the Salicetum alba communities, although it can also develop monospecific stands at intermediate levels between the Salicion softwoods and the hardwood forests.

Survey and Analysis

Field mapping of land use and vegetation type was carried out on a topographic map basis. In a preliminary step the boundaries of the floodplain were derived from a topographic map survey and checked in the field. They coincide with the regularly (once every 10 year) flooded valley floor; irregularly flooded areas in extreme peak events are not included in this analysis.

The field work consisted in the verification of land cover units as they were present in the topographic map. The natural areas of the floodplain and riverbed were mapped for vegetation units. As this mapping was to a more detailed level (patches of minimum 500m^2), the field survey took around 2 weeks for every stretch. For the Meuse the mapping was executed in the summer of 2000, for the Dordogne in 2001. The maps of the Loire and Allier were derived from the project 'Information system on the evolution of the river bed of the river Loire and its tributaries' (SIEL). Within this project a vegetation map was elaborated for the whole river Loire and Allier. Vegetation mapping of the floodplain and river bed of the river Allier and Loire was done in 2000 and 2001 (by the group Mosaïque Environnement, finalised Allier 26/9/2001, Loire 7/11/2001).

Data of the field survey was gathered in spread sheet tables for the 4 stretches. For each patch, the land use, vegetation type and GIS calculated area and perimeter were retained for the analysis.

In a first phase, differences in the riparian landscape for the different stretches were screened, further detailed in the selection of riparian forest patch frequencies and areas. To derive comparable measures of forest development, the riparian forest variables were divided by the stretches length to have a value per running river kilometre.

In the second phase, the river variables for the four stretches were analysed. The hydromorphological pressure parameters and the responses in physical variables and landscape characteristics were compared over the reaches. Finally, the selected river and riparian landscape dynamics measures were related to measures for the riparian forest development, to give indications for the necessary river freedom for viable riparian forest communities. All marked correlations are Spearman correlation test results, obtained in Statistica.

Reference conditions and measures for the riparian landscape and biological integrity

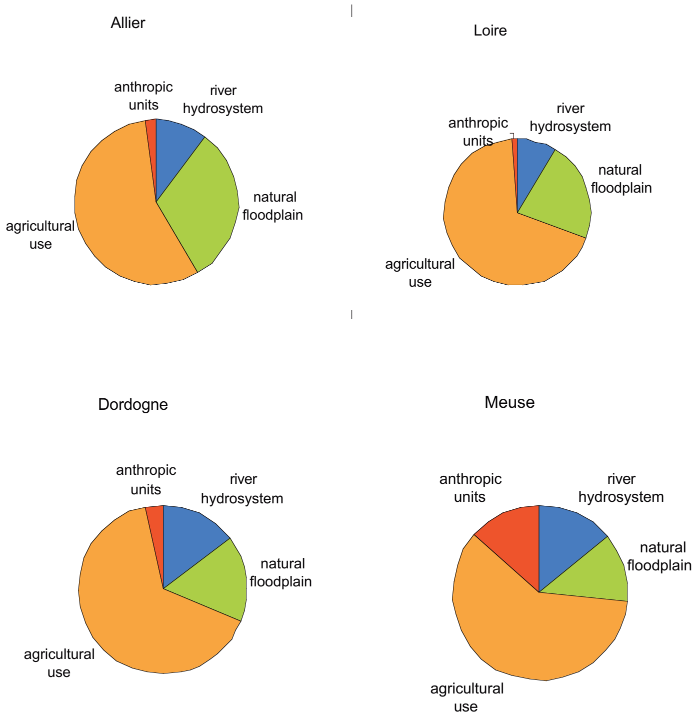


Figure 5.5 Land use of the river-floodplain system for the four studied reaches, classified in 4 categories.

The land use and vegetation survey results, presented in the charts of figure 5.5 (and appendix tables S5), allow a general description and comparison of the land use in the river corridor for the reaches. The Allier reach has more than 31% natural floodplain, the Loire 22%, the Dordogne 16% and the Meuse only counts 12%. For the near-natural river Allier, the natural river dynamics result in large proportions of river bed area with young forest stages and higher rate of Salicetum forest compared to the Loire. Of the Allier's higher floodplain, large parts are in intensive agricultural use. In the Loire valley, as a consequence of the regulation activities (banning of meander and channel migration) more hardwood forests and a large fraction of floodplain meadows (38%) in agricultural use as hayfields are present.

The Dordogne shows a further decrease in natural area, but still has a well developed riparian forest corridor, whereas most of the floodplain is intensively cultivated. Along the Meuse this corridor is absent as a consequence of the total embankment of the reach. The higher rates of running water for the Dordogne and the Meuse are explained by the narrower floodplains, caused by the disruption of parts of the floodplain area by the construction of dikes and for the Dordogne also partly due to the more mountainous/hilly surroundings (see table 5.1).

A measure for riparian landscape dynamics

Riparian landscape dynamics were measured in the rate of open sand, young forest stages of Salicetum purpurea and Salicetum triandrae-viminalis and pioneer vegetation that colonizes the open sediments (Figure 5.6, see also Appendix tables S5). The totals of these categories as percentages of the total floodplain show the dynamics of the riparian landscape for the different reaches. As we can show for the proposed measure the correspondence to the derived measures for hydromorphological pressure (embankment) and physical response (width-depth ratio), we propose this measure as indicator and call it the Riparian Dynamics Indicator (RDI).

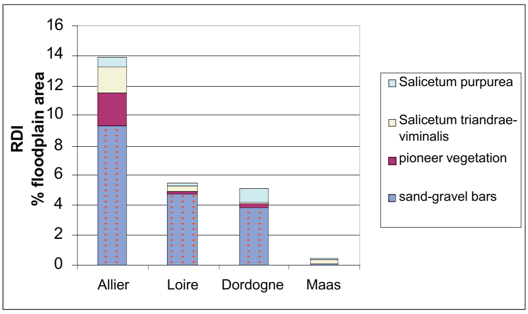


Figure 5.6 Riparian Dynamics Indicator values for the four reaches, composed of survey units in percentage of the river-floodplain system.

The width-depth ratio correlates (r^2: 0.99, p: 0.01) with the riparian landscape dynamics (Figure 5.7d). A less strong correspondence was present with embankment (Figure 5.7a), showing the same grouping for the reaches. For this latter trend an exponential curve fits the data better. This gives an indication of the strong impact embankments can generate, even if they only represent some 10% of the bank length. With just the encroachment/consolidation of some meander bends, which is the case for the Loire and the Dordogne, the river's hydromorphological functioning is highly impacted. The strong responses of the riparian zone can be explained by the changed sedimentological conditions following altered flow conditions and the river incision trend. The linear response of the width-depth ratio offers better opportunities to be used as an indicator for a gradual impact assessment.

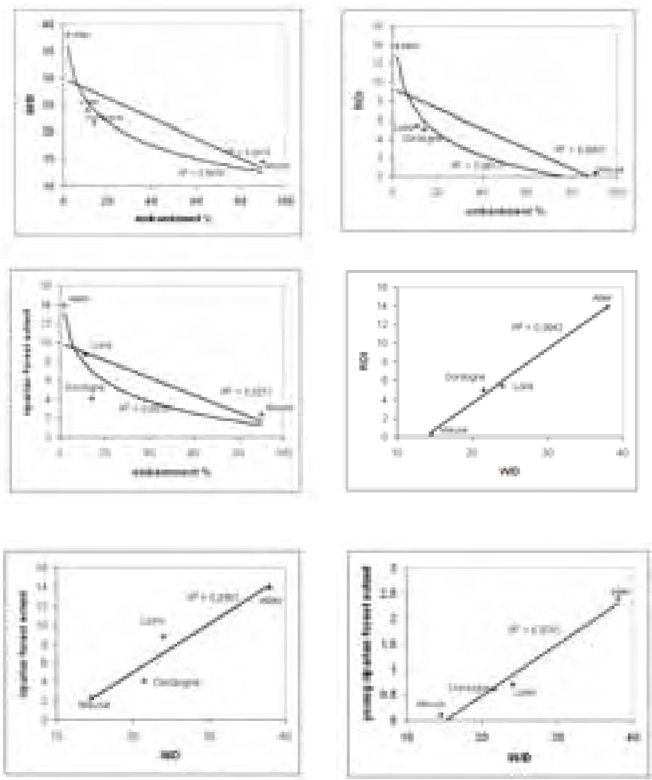


Figure 5.7a–f: Indicators for river hydromorphological alterations: width-depth ratio (a), RDI (b) and riparian forest extent (c) responses to embankment, and the correspondence between the physical geometry (W/D) and landscape (RDI, d) and biotic system (riparian forest e, f) responses are plotted. Linear regression functions with R^2 for Pearson correlation testing are added, for embankment an exponential function is also shown.

Biological quality measures

With the width-depth ratio and RDI as measures for the river freedom and the responses to the hydromorphological pressures, we looked for the best riparian forest development indicator (area, frequency, perimeter, perimeter/area). Area and frequency showed a correlation to the width-depth ratio (r: 0,99, p: 0,013 and r: 0,95, p: 0,049 respectively), perimeter and perimeter/area ratios are not correlated. Especially the estimates of riparian forest area (area per riverkm stretch) for mature (Ripfor) and young phase (RipforY) show good correspondences (Figure 5.7e-f).

The perimeter/area ratio (especially for the young river forest stages) varies also with the dynamics of the river reach, but not significantly (appendix tables S5). With the high dynamics of the Allier, many thin stretches of riparian formations are present. Along the Dordogne and Loire, the riparian stretches of forest are responsible for the high perimeter values, in the floodplains perimeter/area ratios for forest patches decrease strongly, especially in the cultivated Dordogne reach. For the highly fragmented Common Meuse reach, natural patch forms are rare.

The young stages of these gravel river riparian forests (Figure 5.7f) show the high recruitment potential for the Allier. Loire and Dordogne are close to each other in this diagram, which is more conform to the observed trend in hydromorphological conditions (Figure 5.7d). Where for the Loire the recruitment is a little bit less, there is still a large amount of adult poplar forest guaranteeing the survival for the species.

Boundary conditions

The bending point in the embankment curve was above mentioned to correspond to the value of 10% embankment, as critical level for impacts of this hydromorphological pressure. This value corresponds to width-depth ratio 25 and riparian dynamics RDI 8, and for the riparian forest measures Ripfor 6 and RipforY 0,6. This value we retain as boundary condition for a good status, and with the corresponding measures for forest development, we can try to derive riparian forest metrics as ecological quality ratios.

From the near-natural Allier to the heavily modified Common Meuse, with intermediate positions for Loire and Dordogne, the gradual deviation with regard to the little disturbed reference condition is highlighted in the correlation between river freedom, landscape dynamics and the biotic measures of riparian forest area/frequency. It is present in hydromorphological and natural processes of sedimentation and forest development as well as in species. On the species level the presence of the exotic species Box alder (Acer negundo) is of increasing importance in the riparian forests of the Loire and Dordogne. In the near-natural riparian conditions of the Allier, this exotic invading species remains absent. This observation confirms the general rule of invading species taking profit of altered and deteriorated system conditions. High levels of alien species were moreover recorded for the Garonne basin in riparian zones with increased human disturbance, increased habitat fragmentation and a greater departure from natural hydrological patterns (Décamps et al. 1988; Planty-Tabacchi et al. 1996). For the indigenous characteristic species of these communities, the situation is the opposite. Strongest populations of Salix purpurea and Populus nigra are present along the Allier, whereas both species have become almost extinct along the Meuse.

From our analysis, measures and boundary conditions for viable populations and habitat networks for the species can be derived. The presence of a sufficient number of habitat patches is necessary for these species, as recruitment limitation in combination with observed gene flow was recorded their principal threat along gravel rivers (Barsoum 2001, Imbert & Lefèvre 2003, Van Looy et al. 2005a). Along the Allier there is no limitation what so ever, forest patches of all size and age classes are present. This situation is qualified as reference condition. For the Loire and Dordogne situations become more critical. The Loire shows quite a large area of adult riparian forests, but a strongly deviating share of younger stages. The Dordogne comes close to the Meuse for its riparian forest extent, yet, for the Dordogne a minor but still viable population is present, which is indicated by the strong recruitment.

When we determine the rate of the recruitment that survives in the further development, we can determine this critical level. For the Common Meuse, we determined only 10% survival of young phases (Van Looy et al. 2005a), occurring in locations with favourable hydromorphological conditions of width-depth ratio > 25 and morphological activity corresponding to local RDI > 8. For the Loire and Dordogne, only 30% of the young phases was present in locations with these criteria. For the Allier 70% of the young phases is in favourable locations for developing mature riparian forest. For the Loire and the Dordogne, with only 0,6 ha/rkm young forest of which some 70% is expected to fall off, only 0,2 ha/rkm effective regeneration is to be expected. This is surely a critical level and as the recruitment limitation was previously recorded, the measure of the young forest stage can be selected as the most relevant.

Aim of the study was to identify biological measures with potential use as ecological quality ratio. Therefore, from the presented results, measures can be tested for their significance. Proposed measures are: 1) Riparian forest extent (Ripfor) in overall area of riparian forest types/river kilometre (as we concluded for the Allier riparian forest to be high status of development, with a value of 13.9, we can derive an Ecological Quality Ratio score by dividing with 15), 2) Young forest stages (RipforY) in area of young riparian forest stages per river kilometre (values x 4), 3) Target species measure (RipforT) in number of populations (patches) of Popnigra + Salpurp per river kilometre.

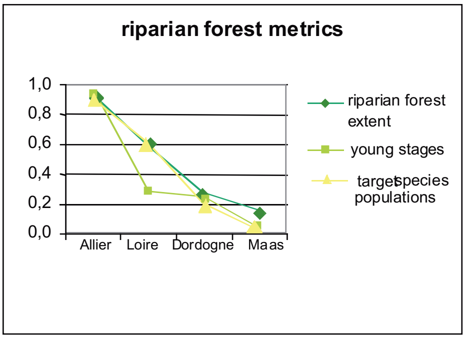


Figure 5.8 Riparian forest metrics derived from the observed biological measures of forest extent of young and mature riparian forest and the patch frequency of target species.

The young stages measure does not show a linear decrease as the other two (Figure 5.8), but the trend conforms more to the other hydromorphological variables for riparian quality we introduced (see Figure 5.7def) - the riparian land use dynamics (RDI) and the width-depth ratio (W/D). As indicator for hydromorphological integrity we propose the young riparian forest stages measure. Young stages and target species' populations attain really low levels in the Meuse reach. This is obvious for a highly regulated reach as we try to identify an indicator for the hydromorphological quality.

Discussion

Reference conditions for hydromorphological and biological aspects of large rivers

The terms 'minor evidence of disturbance' and 'acceptable pressure' have to be applicable for the reference and for the biological quality element at high ecological status. For large regulated rivers like the Meuse, the definition of reference conditions in terms of specific hydromorphological and biological elements is problematic due to a lack of historic data and insufficient knowledge of species-environment relationships. So, important necessary information lacks to define measures for effective ecosystem restoration. The defining of reference conditions can also be based on data of actual references. But actual references for large rivers are also problematic as in the same river basin large river types are unique and between river basins biogeographical differences appear. To what extent river references for large ecoregions can be pointed out and reference conditions derived is an ongoing discussion (Warry & Hanau 1993, Giller 2005). In this paper we tried to follow existing guidance documents to define reference conditions for large rivers. Emphasis was on the relationship between biological elements and the hydromorphological conditions. For specific river types over large geographical areas the best responding groups to the main pressures have to be identified, as different taxonomic groups show different responses to environmental changes and pressures (Heino et al. 2005); riparian forests are in this respect already documented as a key community for the evaluation of hydromorphological alterations in large rivers (Naiman et al 1993, Naiman et al. 1997, Deiller et al 2001, Hughes 2003). However, a hydromorphological assessment is in the WFD only explicitly required for the classification of "high status" and the definition of reference conditions. But we assume that for large rivers, riparian vegetation is an essential component of the "ecosystem structure and functioning" and thus could be included as well in the ecological status evaluation (Gregory et al. 1991, Naiman & Décamps 1997).

We screened four rivers in the same ecoregion, with the little perturbed Allier as candidate reference situation, and the heavily modified Common Meuse at the other extreme. The riparian vegetation we specified in riparian forest types, with qualifying critical species Populus nigra and Salix purpurea. Furthermore we detected temporal sequences of forest development, since the recruitment and settlement of young stages are important aspects with regard to hydromorphological conditions (Schnitzler 1997, Splunder 1998, Van Looy et al. 2005a). As we disposed over a wide range of relevant data for the four selected river reaches, the analysis gave a satisfactory result for pressure responses in measures of landscape and hydromorphological dynamics and in relation to biotic communities, of riparian forest in this case. So, as the Allier showed minimal evidence of disturbance over the investigated parameters, generally we can conclude that the Allier can be used as reference and provides data on reference conditions.

The gradient of disturbance/human alterations to the hydromorphological characteristics over the selected reaches, is clearly measured in the width-depth ratio. The normalisations and embankments of large rivers resulted in incision of the bed (Piégay et al. 2005). The changed channel geometry, for which the width-depth ratio is generally acknowledged as a good descriptor (Raudkivi, 1998), leads to altered hydromorphological conditions. The introduced riparian dynamics indicator (RDI) marks this pressure gradient very well, and conforms to other proposed descriptors for reference conditions of dynamic river reaches (Ward et al. 1999, Middelkoop et al. 2005). The RDI gives a good estimation of the morphological active riparian zone. Including the newly vegetated sediments in this measure is necessitated from practical viewpoint; if we want to use remote sensing data, gathered in late summer as this is the best period for low flow regime, then the sediment zones are covered with ephemeral pioneer vegetation and young tree seedlings. Furthermore the young pioneer vegetation and especially the young riparian forests play an important role in the hydromorphological activity (Hughes et al. 2001, Baptist et al. 2005), in the way that they not only hold substrates, they also induce and promote local sedimentation. As it is a landscape indicator, it bridges the distance between the physical and biological impact of hydromorphological pressures.
The ecological status with regard to hydromorphological conditions proved in our study well-assessed with the riparian forest measures of forest extent for mature and for the young forest stages. As the riparian forests are closely related to the hydromorphological processes of bar and island formation, they are good indicators for the hydromorphological conditions of large gravel rivers (Kollman et al. 1999, Tabacchi et al. 2000, Hughes et al. 2001). The forest area proved the best indicator and gave a good measure (ha/rkm) for riparian forest and its young

phases. Same as Turner et al. (2004), we found patch attributes and edges no strong predictors for riparian forest development.

From the boundary condition analysis, it is clear that not the actual forest extent, but rather the rate of river freedom and riparian dynamics is the critical boundary condition. As we defined a riparian dynamics indicator, the critical boundary condition for riparian forest development is best measured in the RDI value 8. We can also refer to the erodible corridor concept to stress the need for allowing a minimum of riparian dynamics to obtain viable riparian forest communities (Piégay et al. 2005). The aerial photograph of the Allier (Figure 5.4) shows very well what is measured in the RDI.
Where we detected gradual and linear relations between hydromorphological and biological quality indicators, it was the analysis of the target species that revealed the critical boundary conditions.
These conditions nevertheless are best measured in physical variables (Radwell & Kwak 2005).

We can further refer to the correlated river parameter of width-depth ratio, in order to try to derive guidelines for the necessary freedom of the river for the development of riparian forests. Here we find a measure of 25 for the width-depth ratio corresponding to a sufficient riparian dynamics and a sustainable recruitment level, as is present for riparian forest and especially the young riparian forest stage along the Dordogne (Figure 5.7f). This conforms to other studies for reference conditions for the Common Meuse (Van Looy et al. 2005b).

Monitoring proposal

From this research of indicators of hydromorphological and biological quality, we can conclude that the measurement of riparian forest can be restricted to the riparian strip, and so avoiding the difficulty in identification of floodplain borders and the distinction of forest types. The here defined criteria and measures might be useful to derive metrics for the WFD ecological quality assessment, as they showed responses over the whole pressure gradient. Since we only looked at four rivers we could not integrate a statistically relevant set of data to define ecological class boundaries, more data on rivers of this type is necessary to develop WFD proof metrics. Still this kind of metrics offers good perspectives for the integration of hydromorphological and biological responses to human alterations.

A monitoring protocol can be proposed to prove the feasibility and utility of this quality assessment. The derived measures of riparian forest extent, can be measured from remote sensing data (satellite imagery or aerial photography) and simple telemetric techniques. Especially the patch areas can be easily detected with aerial/satellite images. The defined measures (RDI, Ripfor, RipforY) can be derived from an analysis of a digitally delineated riparian corridor. A 50m strip on both sides of the bankfull main channel is proposed in the guidance (CEN 2003). If more detailed information on composition and species populations is required, this needs further control with a field survey. This information can be demanded to evaluate the habitat conservation status for the Habitat Directive (EC 1992), as most of these reaches and their riparian forests are designated protection areas in the Pan-European NATURA2000 network.
In a wider riparian buffer of 5-10x bankfull width, the hydromorphological and biological assessment can be restricted to a screening of larger changes in landscape structures and patterns. These relevant structures could be patches of forest or floodplain channels/oxbow lakes with a frequent survey of changes in shape and detection of signs of erosion-sedimentation processes. With a 6-10 year recurrence period for the survey of a 10 km river stretch, a practical and little expensive evaluation method for this large river type's hydromorphological and riparian vegetation quality can be installed.

Conclusion

A complete understanding of the ecological ramification of river regulation lacks the fundamental knowledge of the complexity and dynamics of intact river systems (Ward et al. 1999). This observation marks the starting point of this research for reference conditions for hydromorphological quality. We first set the picture of hydromorphological modifications and ecological quality referring to the European Water Framework Directive. The gathered hydrological and geomorphological data for the set of four large gravel rivers present in the same ecoregion, together with some information on the alterations during the last centuries, allowed the selection of comparable reaches and the determination of a reference system and reference conditions for the hydromorphological character. The reach of the River Allier proved a good reference and reference conditions were succesfully derived for large gravel rivers of Europe's Western Plains ecoregion. For the hydromorphological reference conditions the width-depth ratio proved the best indicator. The embankment gave important indications to the pressure-impact responses of the large rivers, as we revealed a non-linear, exponential response to this pressure.
The analysis of the biotic community of riparian forests, which correlated significantly to these hydromorphological conditions, resulted in a set of quantitative measures for the reference condition and a monitoring proposal. The measures we derived from the riparian forest analysis showed clear responses to hydromorphological pressures (indicated by embankment and width-depth ratio). The target species analysis showed the critical boundaries and as the set of four rivers spanned the whole range from high to bad ecological status, a measure could be derived applicable for assessment and monitoring purposes in the WFD implementation. The derived measures further allow the evaluation of restoration programmes and conservation efforts for large rivers.

Acknowledgements

We are grateful to the river authorities of France, Belgium and The Netherlands for delivering the data of the gauging stations. The DIREN Loire for approving the use of the SIEL-project data to these purposes and André Chandesris (CEMAGREF Lyon) for the elaboration of GIS-maps. Gertjan Geerling kindly provided the photograph of the River Allier.

V.2
DE DYNAMIEK ONTLEED IN TIJD EN RUIMTE

Van Looy, K., Van Braeckel, A. & De Blust, G. 2005. De dynamiek ontleed in tijd en ruimte. Model voorspelt ontwikkelingen in het Grensmaasgebied. Landschap 22 (3): p126-139

Summary

Model predicts developments in the Common Meuse river.

The ECODYN model is a dynamic model to predict developments in the Common Meuse river system. The model incorporates our understanding of zone and patch structure of vegetation in the river system, governed by flood timing, power and frequency. With these interactions and relationships, a model is build that projects biotic processes over the river system through space and time. With the integration of research results from vegetation succession, forest development and impact of grazing regime at local and regional scale level, a sound modelling approach at ecotope level (scale 1:5000 – 1: 25000) for the river reach was possible. In this article constraints of the model, input data formats and levels and predictive power of the output are described. The modelling results for the Common Meuse restoration project are shown.

Lead

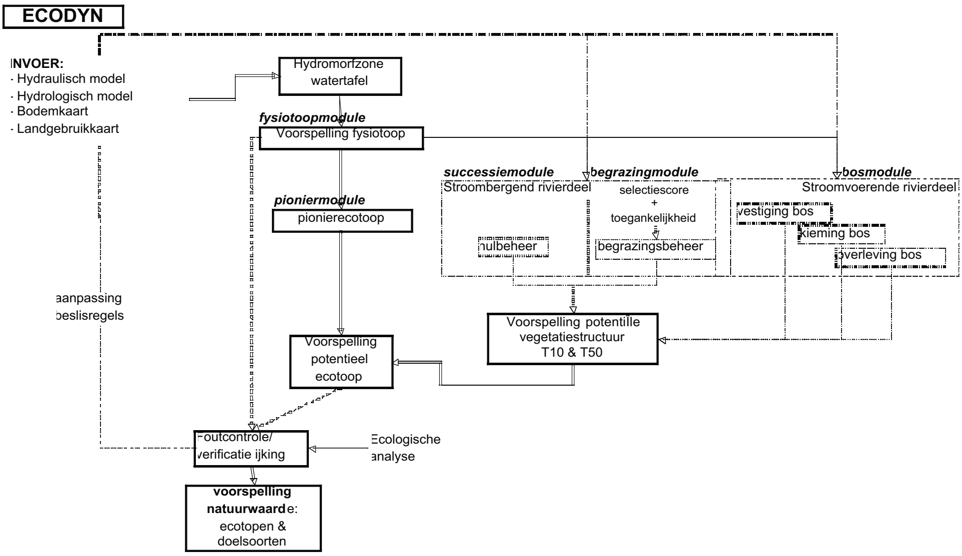
Het ECODYN model is een dynamisch model voor voorspelling van ecotopen in het rivierengebied. Dit model is uitgewerkt voor het Grensmaasgebied. In dit artikel worden de randvoorwaarden, benodigde invoergegevens en betrouwbaarheid van het model besproken. Zowel de eerste validatie als de gemodelleerde structuurontwikkeling en het gegenereerde eindbeeld van het Grensmaasplan, geven een zeer positief resultaat.

De doelstellingen van het rivierenbeheer zijn al enkele jaren in volle beweging. Bevaarbaarheid, hoogwaterbescherming, recreatie, natuurontwikkeling moeten op verschillende plaatsen kunnen samengaan. De realiseerbaarheid daarvan is voor een groot deel afhankelijk van de kenmerken van de rivier en in samenhang daarmee, van de er zich ontwikkelende ecotopen. Debieten, waterstanden en stroomsnelheden, successiestadia en ruimtelijke verdeling van de levensgemeenschappen, vormen zo het natuurlijke kader voor de doelstellingen. Om de kans op het behalen van de doelstellingen van het rivierbeheer in te schatten, moeten de relaties tussen al deze factoren en het verloop ervan door de tijd, bepaald worden. Geïntegreerde, dynamische modellen voor het voorspellen van de invloed van rivierdynamiek en op ecotoopontwikkeling kunnen hierbij helpen om effecten van beheer- en inrichtingvarianten op de natuur te be-

oordelen (Van Kalken & Havno, 1992; Ahn et al.., 2004; Baptist et al., 2004, Haasnoot & Van Der Molen, 2005).
In het Grensmaasproject neemt natuurontwikkeling een belangrijke plaats in (Van Looy & De Blust, 1995; Nagels et al., 1999, Provincie Limburg 2005). Om de mogelijkheden hiervoor zo goed mogelijk in te schatten, is het model ECODYN ontwikkeld. Dit model integreert een aantal systeemprocessen in een expertmodel. Functionele kenmerken van ecotopen en de dynamiek in tijd en ruimte van de Grensmaas, worden erin gecombineerd, zodat een voorspelling van de ruimtelijke ontwikkelingen mogelijk wordt. De directe aanleiding om ECODYN te bouwen, was de weinig betrouwbare output van een vorig model (ecotopengenerator) dat bij de voorbereiding van de Grensmaasplannen gebruikt werd (van Rooij et al., 2000). Daarin was de relatie tussen de ontwikkeling van de vegetatie en de rivierdynamiek onvoldoende uitgewerkt om een ruimtelijke voorspelling te maken van vegetatiestructuren binnen het winter- en zomerbed van de Grensmaas. Er was behoefte aan een instrument dat varianten op het niveau van afgravinghoogte en van de inschakeling van specifieke locaties in het overstroombare deel van de Grensmaas, kon evalueren. Het detail dat hiermee beoogd werd, vereiste een verbeterde afstemming tussen vegetatie- en hydraulische modellering, waarbij vooral de impact van vegetatieontwikkeling op verandering van de ruwheid en riviergedrag een grotere rol zou spelen. Omdat daarnaast ook de effecten van natuurlijke begrazingsvormen in rekening gebracht worden, gaat ECODYN een stap verder dan modellen die de vegetatiedynamiek eenduidig koppelen aan één specifieke milieuvariabele, voor het riviergebied overwegend het hydroregime (Willems 2001, Aggenbach & Pelsma 2003).

Modelconcept

Het ECODYN model werd specifiek ontwikkeld voor het Grensmaasgebied, het ongestuwde traject van de Maas tussen Borgharen en Maaseik. Specifieke gebiedkennis werd opgedaan in een reeks onderzoeksprojecten in de vallei van de Grensmaas (o.a. Vanacker et al., 1998; Severyns et al., 2001; Vanden Broeck et al., 2002; Fourneau et al., 2003; Geilen et al., 2001). In ECODYN worden een aantal modules die ecologische processen beschrijven, stapsgewijs geïntegreerd tot een ruimtelijk voorspellend model (zie figuur 5.9). De opbouw van het model werd opgehangen aan de bestaande rivierkundige en hydrologische modellen die bestaan voor het gebied. Het riviermodel schetst de belangrijkste parameters in een tweedimensionale ruimte; enerzijds de ruimtelijke standplaatskenmerken, anderzijds ook de tijdsstappen in de retourperiode van de maatgevende afvoergolven. De cellen/rasters uit het riviermodel vormen meteen de basiseenheden voor de ecologische modellering. De schaal en nauwkeurigheid van de ecologische voorspelling is rechtstreeks verbonden met deze van het riviermodel. Een overzicht van de gebruikte data en schaal voor de opbouw van het model is gegeven in tabel 5.4.



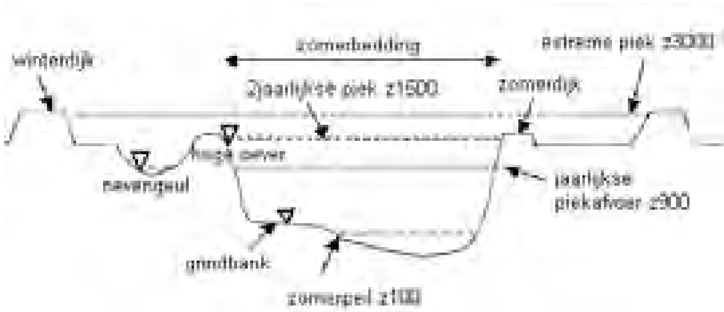
Figuur 5.9 Schematische weergave opbouw ECODYN model.

Figure 5.9 Flowchart of the ECODYN model.

Tabel 5.4 Invoerdata met niveau en schaal voor opmaak ECODYN, en benodigde inputdata voor modeltoepassing (rood).

Table 5.4 Input used for the development of the ECODYN model, with scale and detail level. The input for running the model is in red.

module	Object	Data	niveau	schaal
Fysiotoopmodule	hydrodynamiek	2D-modellering, snelheden, schuifspanningen	Maasvallei	cellen 20-50m
	grondwater dynamiek	GHG, GVG, GLG	deelgebied	25x25m raster
	ecotoop	ecotoopkaart	Maasvallei	min. 25x25m
	hydroregime	retourperioden, Q/H-relatie afvoer-lijn, DTM	Maasvallei	cellen 20-50m
	hydromorfolo-	overlay ecotopen, gische eenheid	Maasvallei stroomsnelheden	cellen 20-50m
Successiemodule	vegetatietype	vegetatieopnamen ecotoopkaart -eenheden	Maasvallei	1x1m/10x10m
	vegetatietype - abiotiek	kartering natuurterrein	deelgebied/	min. 10x10m
	successie vegetatietype	PQ-opnamen	fragmenten	20-50m PQ's
	initiële vegetatie	uitgangssituatie landgebruik, ingreepkaart,	Maasvallei	min. 25x25m
Pioniermodule	hydrodynamiek	2D-modellering, stroomsnelheden	Maasvallei	cellen 20-50m
	hydroregime	retourperioden en Q/H-relatie afvoerlijn	Maasvallei	cellen 20-50m
Bosmodule	bosfase	kartering bomen/ bos rivierbed 1998-2002	17 km riviertraject	grindbank-10x10m
	hydrodynamiek	2D-modellering, schuifspanningen	Maasvallei	cellen 20-50m
	hydroregime	retourperioden en Q/H-relatie afvoerlijn	Maasvallei	cellen 20-50m
Begrazingsmodule	voedselrijkdom	bodemkaart	Maasvallei	min. 25x25m
	grazerselectie-index	vegetatiestructuur-kaart, ruimte-gebruik grazer	natuurterrein	25x25m raster
	Wintertoeganke-lijkheid	grondwatermodel-lering, GHG	Maasvallei	25x25m raster
	ecotoopvorm en isolatie	ecotoopkaart	deelgebied	min. 25x25m
	vegetatiestructuur	uitgangssituatie landgebruik, ingreepkaart,	Maasvallei	min. 25x25m



Figuur 5.10 Schematische voorstelling standplaatsen van zachthoutooibos met kritische afvoer en werkzame schuifspanning (driehoekjes geven standplaatsen waar werkzame schuifspanning berekend werd, de kritische afvoerlijnen zijn aangegeven als z-debiet-waarde).

Figure 5.10 Schematic cross-section of the river bed with critical water levels for the different stages of riparian forest development.

Afbakening van standplaatsfactoren

In de fysiotoopmodule (figuur 5.9) worden op basis van hydromorfologie en grondwater ruimtelijke eenheden afgebakend. Inputgegevens komen uit de rivier- en grondwatermodellering. De afbakening van de hydromorfologische eenheden gebeurt met stroomsnelheidklassen voor rivierbedzones, berekend voor de afvoergolven die maatgevend zijn voor elke rivierbedzone (zie figuur 5.10). De combinatie van stroomsnelheid en overstromingsfrequentie wordt in het grootste deel van het Grensmaasgebied als bepalende factoren voor de standplaats beschouwd (Van Looy & De Blust, 1998; Van Looy et al., 2005). Grondwaterstijghoogten worden berekend aan de hand van periodegemiddelden, in combinatie met bodemkenmerken.
De pioniermodule vormt een verfijning van de fysiotoopmodule en heeft als doel om het periodiek terugzetten van de vegetatieontwikkeling in het model te integreren. Pionierecotopen ontstaan bij een afvoerpiek en de ermee gepaard gaande hogere stroomsnelheden en kennen daardoor een specifieke retourperiode. Ze kunnen zowel in de oeverzone, als in de lage en hoge weerd (d.i. het winterbed van de rivier) voorkomen.

Bij de afbakening werd geopteerd om stroomsnelheden bij stationaire doorrekening van piekafvoeren (d.i. een momentopname bij maximumafvoer) te gebruiken. Om een volledig beeld van erosie- en sedimentatieprocessen bij hoogwaters te krijgen, zou in principe het volledige verloop van de afvoergolf moeten geïnterpreteerd worden. Bij de begrenzing van de stroomsnelheidsklassen werd met deze beperking rekening gehouden door de ruimste grenzen te hanteren op basis van de uitgevoerde ijking in het gebied.

Voorspelling van de vegetatiestructuur per fysiotoop

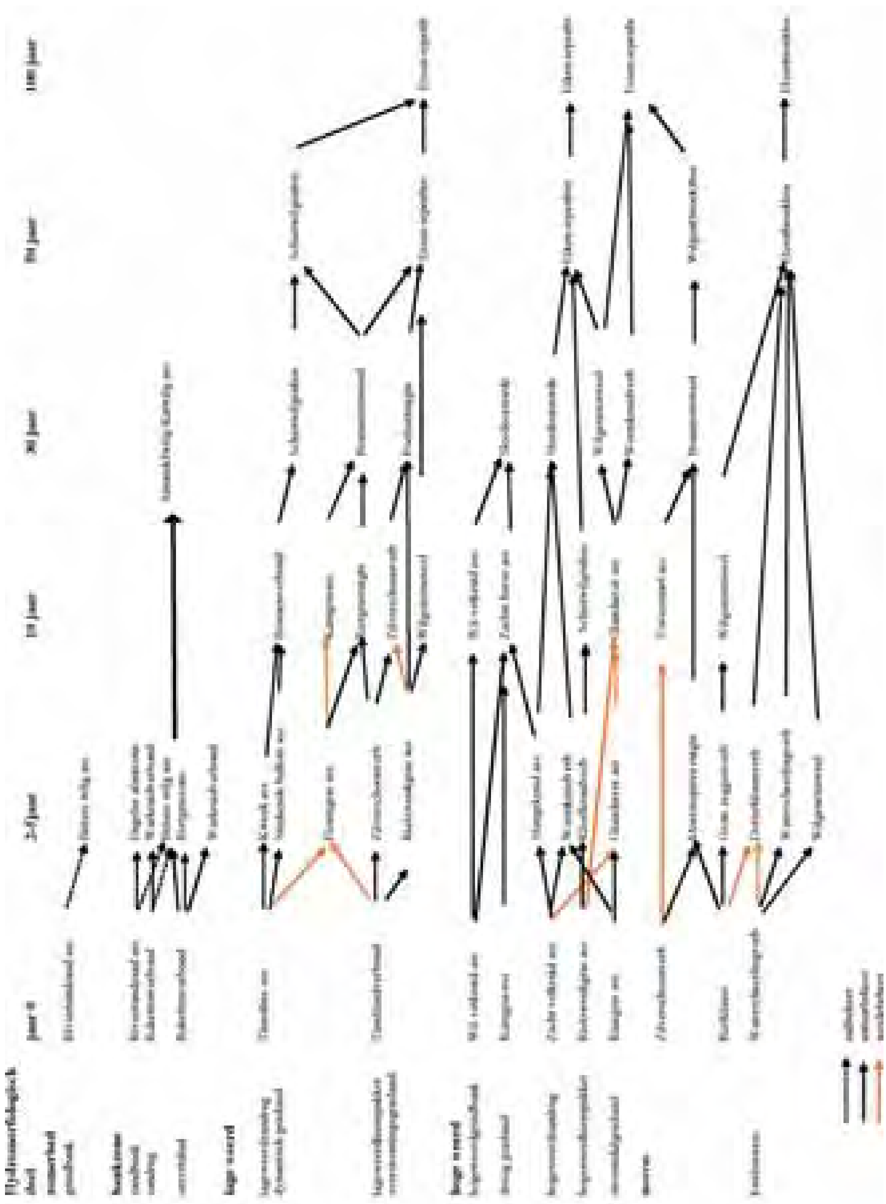
De fysiotopen dienen om het voorkomen van de ecotopen mee te voorspellen. Dat voorkomen wordt uiteraard ook mee bepaald door de soortensamenstelling en het successiestadium van de vegetatie. In de volgende stappen staat de voorspelling van de snelheid en de richting van de successie voorop. Omwille van de verschillen in sturende factoren, wordt een onderscheid gemaakt tussen bosontwikkeling in het stroombergend en in het stroomvoerend gedeelte van de rivier. Als natuurbeheervormen onderscheidt ECODYN een variant zonder beheer en één met natuurlijke extensieve begrazing.
De vegetatieontwikkeling wordt zo opgesplitst in 3 modules:

- Successiemodule die de successie zonder beheer aangeeft in stroombergend en –voerend gedeelte.
- Bosmodule die de bosontwikkeling in het stroomvoerend deel van de rivier voorspelt.
- Begrazingmodule die de potentiële afremming van successie onder invloed van grote grazers in het stroombergend deel van de rivier voorspelt.

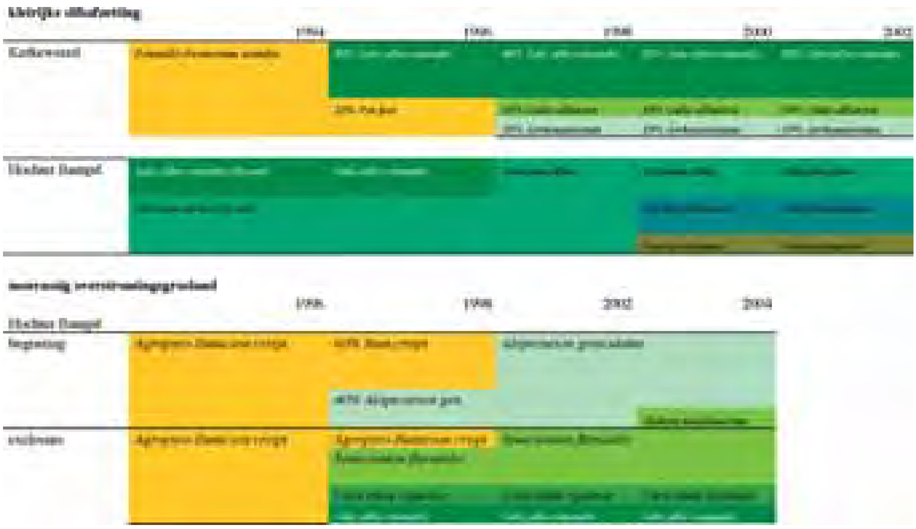
In de successiemodule worden voor de verschillende fysiotopen de structuurklassen pioniervegetatie, grasland, ruigte, struweel en bos onderscheiden en verder onderverdeeld volgens de vegetatietypering (Van Looy & De Blust, 1998). Het successieschema (figuur 5.11) waarin deze vegetatietypen behoren, is afgeleid uit permanent kwadraatonderzoek dat tussen 1996 en 2002 werd uitgevoerd. Dit schema vormt de basis voor de successiemodule en de begrazingmodule en geeft de tijdsstappen (jaar 0-1-2-5-10-20-50) in het model aan.

Figuur 5.11 Successieschema Grensmaasgebied.

Figure 5.11 Succession scheme for the Common Meuse river system.



De successiemodule schetst de ontwikkeling zonder beheer. De invloed van periodieke overstromingen die de successie remmen of vroege successiestadia zoals grindbanken fixeren, zijn in rekening gebracht. Ter illustratie wordt een voorbeeld van successie op een hogeweerd leempakket gegeven gebaseerd op waarnemingen in Kerkeweerd en Hochter Bampd (figuur 5.12). Het leempakket in Kerkeweerd ontstond als open pioniersituatie na het hoogwater van 1993. Na 5 jaar was de open pioniersvegetatie veranderd in een Katwilgstruweel, dat na 10 jaar overging in een Schietwilgenvloedbos. De waarnemingen in Hochter Bampd beslaan de ontwikkelingsfase 10-20 jaar waarbij een Essen-Iepenbos of Elzenrijk wilgenvloedbos gevormd wordt afhankelijk van de overstromingsinvloed (Van Looy et al., 2000). Figuur 5.12 geeft tevens een voorbeeld van een vergelijking van een onbegraasde en begraasde successie, met daarbinnen splitsingen die de ruimtelijke doorvertaling van de successiemodule illustreert.



Figuur 5.12 Successiestappen in enkele permanente kwadraten als input voor de successiemodule voor 2 ecotooptypes.

Figure 5.12 Succession scheme for PQ-plots illustrating the input for the ECODYN succession module for 2 ecotope types.

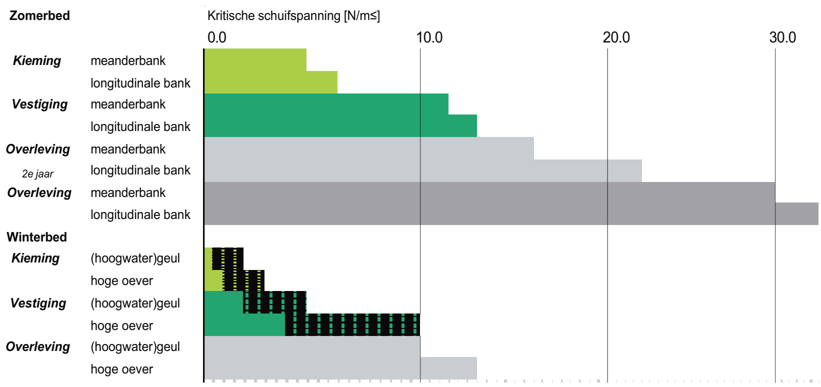
Omdat met de permanente kwadraten niet alle successiefasen gevolgd konden worden, zijn bepaalde vegetatieontwikkelingen ingeschat. Zo zijn onbegraasde situaties in de terreinen met natuurontwikkeling slechts beperkt aanwezig en loopt de ontwikkeling er op de meeste plaatsen nog maar een 10-tal jaar. Uitspraken over de vegetaties die na 30 tot 50 jaar zullen optreden, hebben daardoor een grotere onzekerheid. Daarnaast is ook geen rekening gehouden met verschillen in nutriëntenbeschikbaarheid of initiële soortensamenstelling die binnen eenzelfde fysiotoop kunnen optreden, waardoor eveneens met veralgemeningen gewerkt moet worden.

In de bosmodule wordt de vestiging van zachthoutooibos binnen de stroomvoerende sectie van het rivierbed doorgerekend. Ruimtelijk worden nevengeulen, hoge oevers, longitudinale en meandergrindbanken ('lateral bars' en 'point bars') afgebakend. De tijdfasen in de bosontwikkeling zijn kieming, vestiging (struikfase) en overleving (boomfase). De hydraulische modellering levert de schuifspanningen bij kritische afvoeren, die gebruikt zijn om de mogelijkheden voor bosontwikkeling voor de verschillende tijd- en ruimtesequenties te bepalen.

Schuifspanningen in een model voor ooibosontwikkeling

De korte, heftige afvoerpieken in de Grensmaas, maken dat de overstromingskracht de belangrijkste standplaatsfactor is die de ontwikkeling van rivierbos bepaalt. Jonge bosfasen tot zelfs delen van ontwikkelde bossen kunnen erdoor ontwortelen of omvergeslagen worden. Er is dan ook een duidelijke relatie tussen de werkzame schuifspanningen in de bedding van de rivier en de mogelijkheden voor kieming, vestiging en overleving van zachthoutooibos (Baptist et al., 2005; Van Looy et al., 2005).
Normaal treedt kieming op in beperkte stroken van afzettingen op grindbanken en oevers. Maar zelfs kleinere zomerpieken van 300m^3/s kunnen ervoor zorgen dat kiemplanten op de grindbanken uitspoelen, zodat kieming niet jaarlijks succesvol verloopt. Als kritische afvoer geldt voor de overleving van kiemplanten op de grindbanken dus een gemiddelde zomerpiek, voor de vestiging van jonge bosfasen de gemiddelde jaarlijkse en tweejaarlijkse piekafvoeren (zie figuur 5.10). Voor kieming en vestiging in het winterbed gelden eveneens de jaarlijkse en tweejaarlijkse piekafvoeren als kritisch. Voor de overleving van bos wordt een extreme piekafvoer gehanteerd. Deze afvoeren werden in het 2-dimensionale hydraulische model doorgerekend met als resultaat een schuif-

spanning die de kracht weergeeft die op een specifieke plaats werkt. Deze schuifspanning is het resultaat van de stroomsnelheid en het aanwezige substraat (grof grind voor afgepleisterde grindbanken, grind voor hoge banken, grof zand voor de hoge oever en lemig zand voor hoogwatergeulen).
Voor de in het veld vastgestelde ontwikkelingsfasen en terugzetting van ontwikkeling over de periode 1998-2002, werd in de verschillende zones van het rivierbed (longitudinale bank, meanderbank, hoge oever, nevengeul) de werkzame schuifspanning afgeleid uit de riviermodellering (figuur 5.13). Als resultaat ontstaat een beeld van de ontwikkelingsfasen en de kansen voor ontwikkelend bos over het rivierbed (figuur 5.14).



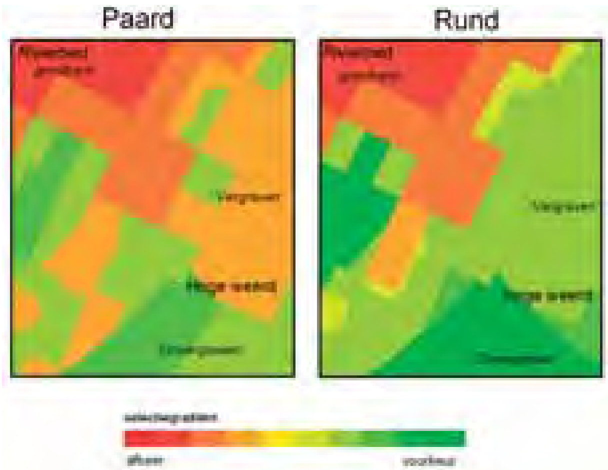
Figuur 5.13 Kritische schuifspanningsranges voor de verschillende ontwikkelingsfasen in de verschillende zones van het rivierbed.

Figure 5.13 Critical shear stress ranges for the different development phases in the different river bed zones.

In de begrazingmodule wordt het effect van een extensieve begrazing op de vegetatiestructuur nagebootst. De basis voor de begrazingsmodule vormen de selectie-index en de graasgevoeligheid per ecotoop, afgeleid uit veldonderzoek (Van Braeckel, 2002; Van Braeckel & Van Looy, 2002). De bepaling van de selectie-index per ecotoop gebeurt aan de hand van indirecte metingen van het terreingebruik nl. mestdichtheid, uitgevoerd in 2001-2003. Bij extensieve begrazing is dit een goede maat gebleken voor zowel rund als paard (Lamoot et al., 2004). Er werden aparte indexen opgesteld voor de twee grazertypes (zie figuur 5.15). De ruimtelijke spreiding van de graasintensiteit wordt tenslotte verkregen door de ecotoopspecifieke selectie-index te corrigeren voor wintertoegankelijkheid, isolatie en plekgrootte. Het resultaat is een relatieve maat voor graasintensiteit. De graasgevoeligheid, de maat voor de afremming van successie, wordt voornamelijk bepaald door de initiële toestand met betrekking tot de vegetatiestructuur, het bodemtype en vochtigheidsgraad,die we afleiden uit de vegetatiekaart, de bodemkaart en de fysiotoopmodule.

Figuur 5.14 Resultaat bosmodule met weergave van de verschillende ontwikkelingsfasen zoals voorspeld met ECODYN.

Figure 5.14 Result of ECODYN forest module showing the model outcome for the different forest development phases.



Figuur 5.15 Illustratie van toegekende selectie-indexen voor paard en rund over een deel van het riviergebied.

Figure 5.15 Illustration of selection indices for horse and cattle in a small part of the river bed.

Figuur 5.16 Pilootproject Meers met zicht op grindbank en lageweerdruigte (met rijtje vestigende wilgen op overgang) en hogerop struweelontwikkeling.

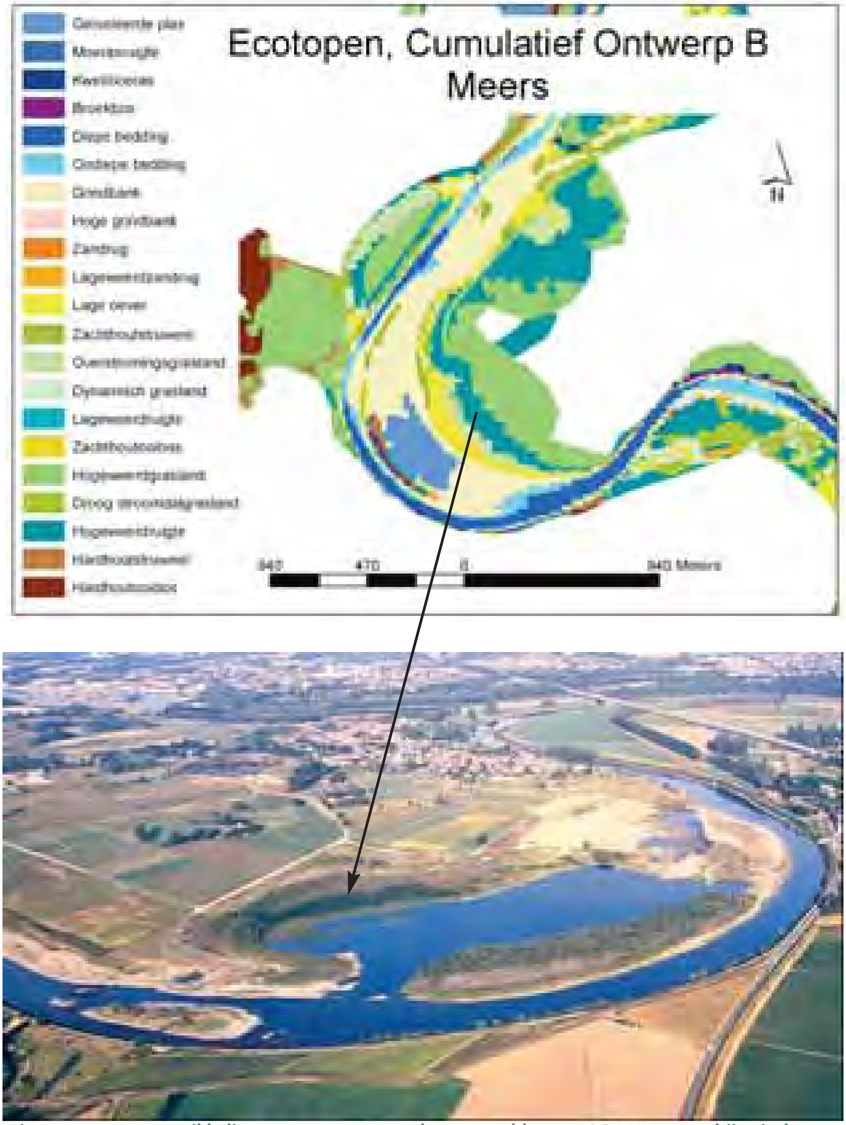
Figuur 5.16 View of the pilot project at Meers, with the gravel bar, lower floodplain herbaceous vegetation and forest development.

De combinatie van graasintensiteit en graasgevoeligheid leidt voor elk ecotoop in het gebied tot een specifieke fixatie, vertragen of onbeïnvloed laten van de successie. Dit ruimtelijk patroon van begrazingsinvloed wijzigt in de successiestappen, zodat een iteratieve module gecreëerd is die elke tijdsstap doorloopt.

Validatie van het model

Een gebiedsdekkende validatie is onmogelijk gezien het overwegend landbouwkundige gebruik van het gebied. De huidige natuurterreinen in het gebied die een validatie zouden toelaten, liggen tevens overwegend in verstoorde milieus (heraangevulde grindwinningen achter hoge zomerdijken). Op kleine schaal is een beoordeling wel mogelijk:

Het proefproject van Meers geeft na 8 jaar een beeld van de ontwikkelingen in de dynamische zones van grindbanken en lage oevers. Het beeld dat ECODYN genereert van 10 jaar ontwikkeling, toont een vergelijkbare ruimtelijke ecotoopbegrenzing als in het veld of op luchtfoto af te leiden valt (figuur 5.17). De ecotopen die voorspeld worden komen ook goed overeen met de waarnemingen in het terrein. Op het eiland tegen de rivier zijn hoge grindbanken en zandruggen aanwezig op de stroomopwaartse kop van het eiland. Verderop is er de overgang vanaf de ondiepe bedding naar de grindbank, de lageweerdruigte en het ontwikkelend struweel bovenop het eiland (figuur 5.16). Aan de landzijde van de plas ontwikkelt lage oeverruigte gedomineerd door Beklierde duizendknoop (Polygonum lapathifolium) en op luwe zones zachthoutstruweel, hogerop opgevolgd door ruig overstromingsgrasland met Fioringras (Agrostis stolonifera) en Rietgras (Phalaris arundinacea) en verder overgaand in Bijvoet-hogeweerdruigte.



Figuur 5.17 De ontwikkeling van ecotopen zoals voorspeld met ECODYN, verschijnt in het pilootproject Meers duidelijk op het terrein.

Figure 5.17 ECODYN output and areal view of the pilot project at Meers.

De hogeweerdgrindbank en hogeweerdzandrug zijn dynamische afzettingsmilieus op de hoge weerd die omwille van hun hoge natuurwaarde en specifieke tijd- en ruimtegebondenheid een interessant validatie-object vormen. Ze ontstaan bij hogere afvoergolven, waarbij de storingsinvloed van de zomerdijken niet meer van tel is. Deze pionierecotopen werden gekarteerd en geïnventariseerd na elk hoogwater voor de periode 1994-2002. De verspreiding van de kensoorten van deze ecotopen werd voor de volledige Maasvallei nagegaan. Kensoorten voor de hogeweerdgrind-zandruggen (Van Looy, 2000) zijn ook vaak kensoorten van de droge stroomdalgraslanden, het verdere successiestadium van deze pionierecotopen (Jansen & Schaminée, 2003). In een eerste stap werden de voorspelde plekken in het veld geïnventariseerd of ze effectief dat pionierecotoop vormen. In een tweede stap is voor de kensoorten met gekende verspreiding nagegaan in hoeverre ze ook binnen de voorspelde plekken te vinden zijn.
Tabel 5.5 toont het huidige beperkte voorkomen van deze ecotopen, slechts 12% van de oppervlakte, in vergelijking met de voorspelling (model). Wanneer we het aantal plekken bekijken waar in de veldinventarisatie nog kensoorten van de habitat aangetroffen werden, blijkt een groter overeenkomst (48%). Een selectie van de gemodelleerde plekken onder weiland/hooiland en natuurareaal werd gemaakt omdat daar het landgebruik de ecotoopontwikkeling zou kunnen toelaten. Deze plekken tonen inderdaad 85% overeenstemming met de veldwaarneming van kensoorten (27/32 plekken). De grote oppervlakte in het model toont dus wel degelijk de grote potentie voor het ontwikkelen van deze ecotopen langs de Grensmaas. De afwijking in huidige oppervlakte is grotendeels te wijten aan het intensief landbouwgebruik van het gebied, waarbij hoge grind-zandafzettingen vlot worden genivelleerd en ingeploegd. De kensoorten werden dan ook vaak enkel op perceelsranden aangetroffen.

Tabel 5.5 Oppervlakte en frequentie van gemodelleerde en gekarteerde pionierecotopen
Table 5.5. Surface and frequency of modelled versus field mapped pioneer ecotopes.

PIONIERPLEK	Oppervlakte (ha) model	Oppervlakte (ha) veld 2000	Frequentie van plekken model	Frequentie van plekken model selectie	Frequentie van plekken Veld met kensoorten
Hogeweerdgrindbank	44	11	23	12	10
Hogeweerdzandrug	114	8	33	20	17

Tabel 5.6 Voorspelde aandeel kensoorten gekarteerd in het gebied

Table 5.6 Percentage of predicted patches of typical species for the pioneer ecotopes, surveyed over the river reach.

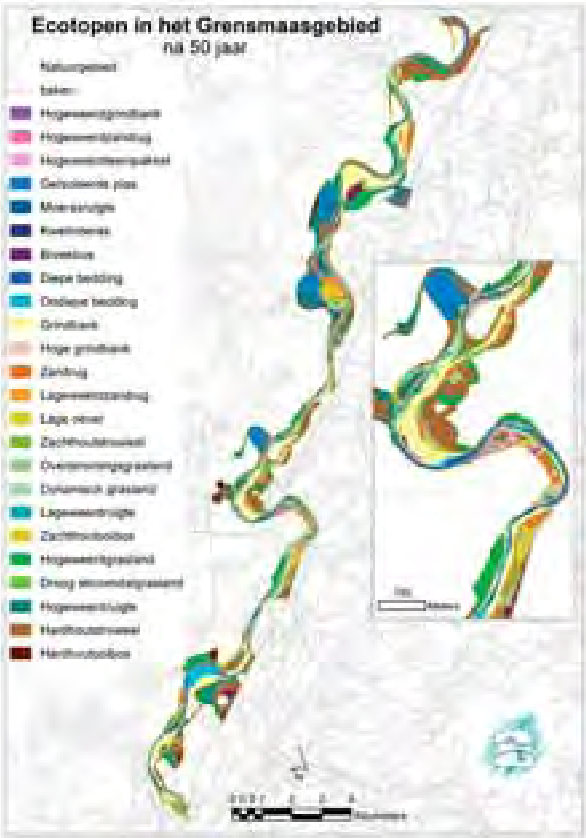
Ecotoop	Kensoorten	Wetenschappelijke naam	# plekken	% voorspeld
Hogeweerdgrindbank	Wit vetkruid	*Sedum album*	9	78
	Muurpeper	*Sedum acre*	14	71
	Tripmadam	*Sedum reflexum*	3	100
	Ronde ooievaarsbek	*Geranium rotundifolium*	6	100
	Steenhoornbloem	*Cerastium pumilum*	6	66
	Eironde leeuwebek	*Kickxia spuria*	3	33
Hogeweerdzandrug	Grote tijm	*Thymus pulegioides*	2	100
	Zacht vetkruid	*Sedum sexangulare*	6	66
	Kandelaartje	*Saxifraga tridactylites*	4	50
	Rozetkruidkers	*Lepidium heterophylum*	2	50
	Plat beemdgras	*Poa compressa*	8	87
	Sikkelklaver	*Medicago falcata*	5	100
	Veldsalie	*Salvia pratensis*	5	80
	Wondklaver	*Anthyllis vulneraria*	2	100

Ook in de omgekeerde validatie-oefening voor het voorkomen van de specifieke kensoorten in gemodelleerde plekken blijkt de voorspelling vrij goed te zijn (tabel 5.6). Voor de soorten die enkel dicht bij de rivier aanwezig zijn zoals Grote tijm (Thymus pulegioides), Sikkelklaver (Medicago falcata), Ronde ooievaarsbek (Geranium rotundifolium), Wondklaver (Anthyllis vulneraria) en Veldsalie (Salvia pratensis), liggen de scores zeer hoog. Soorten zoals Eironde leeuwebek (Kickxia spuria) en Kandelaartje (Saxifraga tridactylites) scoren lager aangezien een aantal standplaatsen ontstaan zijn door grindwinning, en dus niet door het riviermodel voorspeld kunnen worden.

Toepassing

De toepassing van ECODYN voor het Cumulatieve Vlaams-Nederlandse Ontwerp voor de Grensmaas toont de mogelijkheden van het model (Van Braeckel & Van Looy, 2004). De mate van detail van de resultaten maakt een uitgebreide evaluatie mogelijk. De voorspelde ecotoopverdeling stemt goed overeen ten aanzien van het ecologische toetsingskader (Helmer & Klink, 1995) met eerdere inschattingen

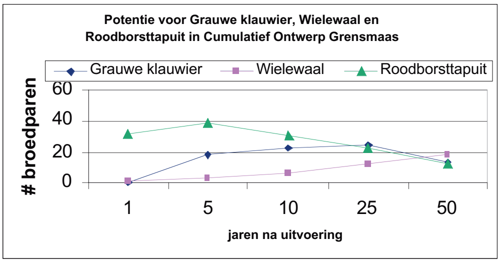
(Peters & Hoogerwerf, 2003). Er ontstaat een beeld van het riviergebied met een grote variatie aan ecotopen in het dynamische deel van de rivier en een grotere uniformiteit in de hoge weerden (figuur 5.18).



Figuur 5.18 ECODYN ecotoopvoorspelling voor het Vlaams-Nederlandse Grensmaasplan.

Figure 5.18 ECODYN result for the Common Meuse restoration project after 50 years.

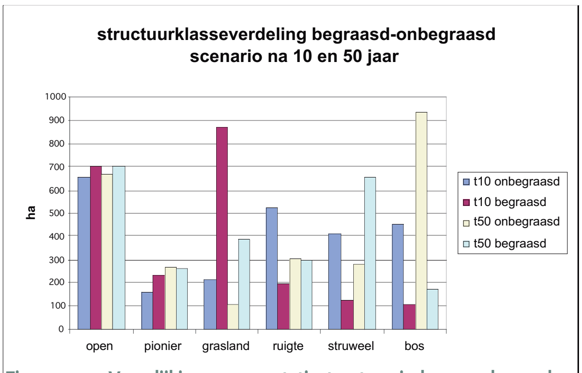
Naast een evaluatie ten opzichte van het vastgestelde ecologische toetsingskader, is een uitgebreide beoordeling op basis van doelsoorten (cfr. Duel et al. 1996) toegepast. De doelsoorten tonen het tijdsfacet van de successie en geleidelijke opbouw van een evenwicht tussen rivierpioniers en grasland- en bossoorten. Dit kan geïllustreerd aan de hand van enkele vogelsoorten die doorheen de ontwikkeling van 50 jaar na uitvoering van het Grensmaasproject de trends in de vegetatiestructuur volgen (figuur 5.19).



Figuur 5.19 Potentie voor een aantal broedvogelsoorten in de tijd bij uitvoering van het Grensmaasplan.

Figure 5.19 Potential breeding population of bird target species over a 50 year time span after river restoration.

De voorspelde ontwikkeling van bos en de terugzetting van successies in het riviergebied in tijd en ruimte zijn aspecten die in andere modelleringen veelal ontbreken. In het onbegraasde scenario voorspelt het model na 10 jaar 25% en na 50 jaar meer dan 50 % bos in het gebied (figuur 5.20). Met natuurlijke begrazing geraakt ook wel een derde van het gebied bebost na 50 jaar. De voorspelde structuurontwikkeling toont bij doorrekening van de ruwheid in het hydraulische model bij een maatgevende afvoerpiek een opmerkelijke daling van hoogwaterstanden in vergelijking met de voorheen gehanteerde vegetatieruwheid (een gemiddelde daling van 9cm over het gehele traject werd voorspeld!). Dit resultaat wordt toegeschreven aan de meer aanvaardbare voorspelde positie en vorm van ecotopen met hoge stroomweerstand. Het bevestigde de problematische, niet-accurate voorspelling van de 'random' ecotopengenerator.



Figuur 5.20 Vergelijking van vegetatiestructuur in begraasd en onbegraasd scenario in de tijd.

Figure 5.20 Vegetation structure repartitions in grazed and ungrazed conditions for different time span (10, 50 year) after river restoration.

Beperkingen

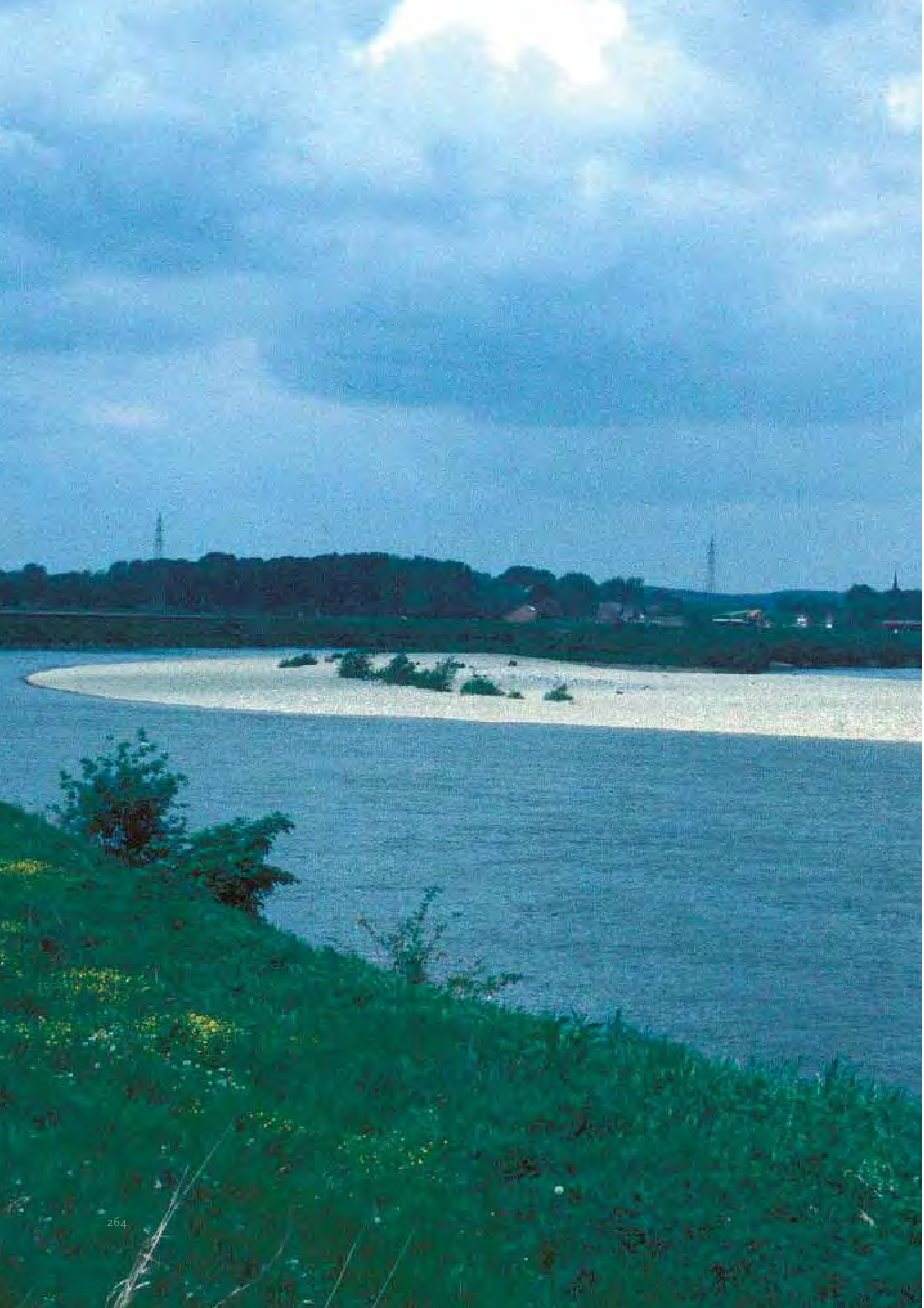
De betrouwbaarheid en informatie van een modeluitkomst hangt nauw samen met de invoer. Aangezien we voor de opmaak van het model over uitgebreide, gedetailleerde informatie van het gebied beschikten (tabel 5.4), kunnen we ook voor de voorspelling doordringen tot op het schaalniveau van het ecotoop (1:5.000-1:25.000)(Klijn,1994). Dit is het locale niveau van projecten en rivierherstelmaatregelen dat we beogen om varianten in afgravingsniveau en plaats van oeveringrepen te kunnen evalueren.
Beperkingen in betrouwbaarheid in de huidige vorm zijn de temporele aannamen die gebruikt worden binnen de successie- en begrazingsmodule. Hier is slechts beperkte informatie voorhanden waardoor de voorspelde ontwikkeling verder in de tijd onbetrouwbaarder wordt, anderzijds ook doordat uitgegaan wordt van de hydrodynamische berekening zonder rekening te houden met een veranderende morfologie. Goede morfologische modellen die de ontwikkeling van het terrein kunnen schetsen, zouden dan ook een waardevolle aanrijking vormen om de ecotoopvoorspelling in de tijd meer betrouwbaarheid te geven. Helaas blijken deze modellen voor riviersystemen met een gegradeerde bedding nog voor grote problemen te staan (Akkerman, 2003).

De beperking van de voorspelling is dus enerzijds afhankelijk van de nauwkeurigheid en mate van detail van de invoer, anderzijds is de stochasticiteit van het rivierregime een grote boosdoener om voorspellingen naar plaats en tijd te doen in het riviergebied. De afgelopen 10 jaar kreeg de Maas een 10-tal hoogwaters te verwerken met retourperiode 20 jaar. Desondanks werkten we bij deze modellering met een gemiddelde afvoertijdsreeks op basis van retourperioden, waarmee het resultaat dus een generaliserend beeld geeft van de ontwikkelingen en variatie doorheen het gebied in de tijd.

Conclusies

Met ECODYN volgen we de keuzen en opties die in zwang zijn voor het opmaken van modellen. Olff et al. (1995) pleitten al voor een meer dynamische aanpak in de expertsystemen waarin tot op heden overwegend statische correlaties toegepast worden. Wassen & Verhoeven (2003) onderstrepen tevens de kracht van specifieke modellen, aangezien modeloplossingen voor specifieke problemen, bruikbare elementen kunnen aanleveren voor complexere modellen (Van Oene et al., 2000). Scheffer & Beets (1995) verkiezen bovendien pragmatische benaderingen gebaseerd op eenvoudige empirische relaties aangevuld met expertkennis, eerder dan complexe simulatiemodellen van ecosysteemprocessen. Brede, multidisciplinaire modelbenaderingen worden tot slot zeer belangrijk geacht in het evalueren van doelstellingen van complexe planprocessen zoals rivierherstelprojecten (Van den Bergh et al. 2005; Watanabe et al. 2005). De opmaak van ECODYN trachtte gehoor te geven aan deze oproepen.

De resultaten van de toepassing van ECODYN zijn zeer bemoedigend. De met ECODYN gemodelleerde ecotoopgrenzen zien we in nieuwe ontwikkelingen zoals in het proefproject Meers mooi opkomen in het terrein. Ook de met ECODYN geschetste bosontwikkeling gaf een veel betrouwbaarder beeld dan de voorheen met een ecotopengenerator gecreëerde voorspelling. De resultaten voor het Grensmaasproject pakten positief uit voor de ruwheidsdoorrekening en hoogwaterberekeningen en gaven nieuw perspectief aan het vraagstuk van stroomweerstand en natuurontwikkeling. Het gaf tevens een bijkomende stimulans om met meer accurate modelleringen te gaan werken voor het verdere Planontwerp. Momenteel wordt het model verder verfijnd voor de doorrekening van locale projecten, waarbij ingrepen en varianten gedetailleerd in de modellen worden ingebracht. Met de verfijnde modelvorm en meer gedetailleerde invoergegevens worden de komende jaren de Grensmaas-ingrepen over beperkte deelgebieden gemodelleerd. Verbeteringen zijn voorzien in de begrazingsmodule en de successiemodule vanuit validering aan de hand van de terreincampagnes van de afgelopen jaren en de monitoring van pilootprojecten (Van Looy 2005). De toepassing van ECODYN-varianten op andere rivier(traject)en of natuurontwikkelingsprojecten (bv. vergravingen) behoort eveneens tot de mogelijkheden.



V.3

INTEGRATION OF ECOLOGICAL ASPECTS IN FLOOD PROTECTION STRATEGIES: DEFINING AN ECOLOGICAL MINIMUM

Geilen, N., H. Jochems, H., Krebs, L., Muller, S., Pedroli, B., Van der Sluis, T., Van Looy, K. & Van Rooij, S. 2004. Integration of ecological aspects in flood protection strategies: defining an ecological minimum. River Research and Applications 20: 269-283 .

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Abstract

Policy makers are confronted with the question how to combine sustainable flood protection and floodplain rehabilitation in the best possible way. Both topics deal with spatial planning aspects in a range of scales. This question was the starting point for the development of an evaluation method within the IRMA/SPONGE project INTERMEUSE, illustrated on the basis of assumed flood protection strategies in the Meuse river basin (the "sponge" strategy, the "retention" strategy, and the "floodplain lowering" strategy). The integration of flood protection and floodplain rehabilitation can be performed on two scale levels that are interrelated: on the regional level the focus is on (large parts of) an entire stream basin, on the local level specific site conditions are taken as starting point. Ecological aspects under study are spatial cohesion of habitats as identified by species population persistence modelling (regional, longitudinal level) and required habitat quality for carabid beetles and for meadow vegetation gradients as assessed by correspondence analysis (local, transversal level). The carabid beetles are taken as indicative for the ecological integrity of the river bed, the meadow vegetation for that of the floodplain.

Unifying concept in the evaluation of ecological integrity is the ecological minimum: the critical boundary or minimum level of habitat conditions for a potentially good ecological functioning. It is the least acceptable state for a river ecosystem that is still functional to some extent, compared to a natural river ecosystem. The results of this study show clearly that there is a good chance to combine floodplain rehabilitation aims with flood protection activities, both on a local and on an international scale. Although ecological effect assessment and ecological optimising (referring to a natural reference state) remain basic, additionally the assessment of the ecological minimum helps defining design strategies for integrated flood protection, especially in situations where river rehabilitation is an opportunity.

Introduction

The natural river landscapes in NW Europe have changed drastically over the last centuries due to human activities. Normalisation and regulation of the river ensured quick run off of water, ice and sediments and at the same time enhanced navigation. Dikes were raised to protect people and goods from flooding. The remaining floodplain areas are almost completely in use for agriculture and at some places gravel, sand or clay mining has been carried out (Van Dijk et al., 1995). The massive flooding events of 1993 and 1995 along the rivers Meuse and Rhine and of 2002 along the Elbe demonstrated that the presumed safety against flooding is to be reconsidered.

In the past dikes were raised after (potential) flood events; now it is clear that new strategies need to be developed as further raising of dikes is not a long term solution (Van der Kraats, 1994). The central theme of these new strategies is to give back the rivers some of the "room" they had lost in the past centuries (Pedroli & Postma, 1998). However, space is scarce and this is especially true along and around river systems. Apart from flood protection other river functions claim the scarce available space, like urbanisation, industry, recreation, agriculture and nature (Lorenz et al., 1997). Therefore, to realise the new strategies in flood risk management, so-called 'win-win' situations need to be achieved, i.e. measures that are beneficial for various river functions. Several functions, e.g. nature, could benefit from the changes in river management that will take place to maintain flood protection.

Natural features of river systems are the result of dynamic geomorphological processes (Wolfert et al., 2001). As a result of the above mentioned human activities the impact of these processes diminished and the natural river landscape deteriorated. With the decline of natural habitat diversity, the accompanying characteristic species vanished or were left in isolated scattered fragments of habitats. The last decades national and international programs have started aiming at the ecological rehabilitation of river systems. The guiding principle for this needs to be the (restoration of) natural river processes: in particular the hydro- and morphodynamics. Concomitant with the expected large scale changes in spatial design of floodplain areas along NW European river systems, resulting from flood protection measures, tuning of measures and aims for the ecological rehabilitation of river systems have become a prerequisite.

The translation of new flood protection strategies into daily practice incorporating ecological rehabilitation goals, calls for new concepts and accompanying tools

which can help the stakeholders to explore future spatial designs for floodplain areas (Smits et al., 2000). Both flood protection and river rehabilitation are strongly served by an integrated approach on a river basin level, partly as space is scarce, partly as problems cannot always be solved at the particular site in question. For both flood protection and river rehabilitation it is not enough to have sufficient space, also a good spatial connectivity is important, even a necessity. For flood protection this coherence is even the guiding principle for future spatial arrangement. The same applies for conservation and restoration of natural assets.
This paper is one of the outcomes of the project INTERMEUSE under the IRMA/SPONGE umbrella, directed to the development and application of a methodology for the evaluation of spatial planning alternatives for river basins, with respect to the integration of flood protection and floodplain rehabilitation. Focussed is on the case of the river Meuse. For a complete description of the INTERMEUSE project is referred to Geilen et al. (2001).

Principles of river restoration

Integrating flood protection and river ecology

Integration of flood protection and floodplain rehabilitation focuses on the following process: in order to maintain safety against flooding a certain flood protection measure (or strategy) will be carried out, resulting in changes in the abiotic environment that in turn will influence biological succession and potential. To integrate the goals of both flood protection and floodplain rehabilitation, knowledge on this basic theme and understanding of the interrelations is of utmost importance.
As stated, the central theme in modern flood management concepts is to give back the rivers some of the "room" they had lost in the past centuries. In the Netherlands this concept has lead to a new policy directive "Room for the River" (Anonymous, 1997). The new strategies for flood protection concentrate on the following principles:

- Retaining water to slow down run-off to the main streambed and thus lowering the peak discharge. In practice this strategy applies mostly to upstream reaches and tributaries and may consist of land-use changes or re-meandering of streams;

- Retention of peak discharges. In practice this strategy applies mostly to the upstream parts of the main river bed and can be performed inside the winter bed or outside in specially designed “retention basins”;
- Increasing discharge capacity to ensure quick run-off of water. In practice this can be performed for example by floodplain lowering or river bed widening.

The ecological effects of flood protection measures related to one of the above principles have been assessed, using a scenario approach. Based on the above principles three distinct flood protection strategies were stated: ‘Sponge’, ‘Retention’ and ‘Winter bed’ (i.e. floodplain lowering). Due to its characteristics each strategy will result in specific ecological potentials differing in scale and type, thus creating different chances to integrate flood protection and river rehabilitation goals.

Longitudinal and transversal aspects

In most lowland rivers, flood protection will dominate the process of decision making in river management. Integration of flood protection and river rehabilitation will focus on optimising the future situation on the basis of river rehabilitation demands. Because ecological integrity and biodiversity patterns are scale-sensitive (Wiens, 1989; Ward et al., 2002), regional comparisons cannot be applied to local scales. Thus, integration should focus on mutual aspects at different scale levels. Within the project INTERMEUSE this is elaborated for two scale levels.
On the scale of river basins longitudinal aspects form the basis for the integration of flood protection and floodplain rehabilitation goals. For flood risk management scale and configuration of measures determine the impact and sustainability of flood protection strategies. The same is true for river ecosystem quality at this scale level as is elaborated in the river continuum concept (Vannote et al., 1980), one of the theoretical concepts for river rehabilitation. In this project, ecological network analysis of habitat configurations is used to assess the impact of the flood protection strategies on the development of viable populations of species as indication of the river ecosystem integrity.
On the scale of floodplains we assume that completeness of transversal gradients form the basis for integration. As river ecosystem quality at this scale level is large-

ly determined by the impact of dynamic abiotic processes, the presence of gradients is an important prerequisite. This constitutes the basis of the flood pulse concept (Junk et al., 1989), another important theoretical concept for river rehabilitation. There is a direct linkage with flood protection through the design and dimensions of physical planning measures.
The transversal aspects focus on species assemblages in relation to local conditions, as indication for ecological quality. In this project, carabid beetles were chosen as indicator group for the river bank, and floodplain meadow vegetations for the floodplain. From the ordination of data and the correlation with groupings of environmental variables, predictor variables of river conditions for the biota can be quantified (Petts & Bradley, 1997). The tolerance of species (groups) to habitat conditions allows the quantification of boundary conditions based on species or communities at risk (Hansen et al., 1999).

The ecological minimum as a design parameter

Traditionally the assessment of river ecosystem quality has been based solely on the measurement of physical, chemical and some biological characteristics. These measurements are not very useful for large-scale management of catchments or for assessing whether river ecosystems should be protected or not (Fairweather, 1999; Norris & Thoms, 1999). New approaches try to combine as many ecosystem indicators as possible, based on relationships between environmental variables and biota in the river system (Petts et al. 1995; Petts & Bradley 1997). In many publications the number and size of patches of streambed and riparian communities and the presence of suitable habitat for threatened species are proposed as criterion in the evaluation of rehabilitation and protection needs (Van Kalken & Havno, 1992; Reijnen et al., 1995; Lamouroux et al., 1998; Hansen et al., 1999; Palmer et al., 2000; Vis et al., 2001). The principle element for the integration of flood protection and floodplain rehabilitation as it is elaborated in this project, is the identification and quantification of key elements, to incorporate floodplain rehabilitation aspects in spatial planning and integrated effect assessment. Starting point is the identification and quantification of the so-called “ecological minimum”, the critical boundary or minimum level of habitat conditions for a potentially good ecological functioning.

According to the new EU Water Framework Directive, all rivers should obtain at least a “good ecological status” (European Union, 2000). In defining ecosystem health, the “good ecological status” needs to be quantifiable, based on knowledge of species and community responses to natural processes and human pressures (Karr, 1999). Comparison of current conditions to desired post-restoration conditions determines the relative “health” of the system, with the possibility to define minimum values falling within the desired range of values of a good health (Kershner, 1997; Norris & Thoms, 1999). The ecological minimum as used in the INTERMEUSE project is a critical level of habitat availability corresponding with the lowest acceptable level of ecosystem functioning (Karr, 1999). This is elaborated for the longitudinal and transversal aspects mentioned above, on the basis of the results of the ecological effect assessment for the proposed flood protection strategies.

Regional integrity: networks of viable populations

Method

On a regional scale, spatial planning alternatives can be assessed on potentials for ecological integrity by means of a habitat network analysis (Pedroli et al., 2002). The ecological rehabilitation goals and therefore the analysis focus on the spatial configuration of habitats. A number of habitats within reach of each other can form an ecological network, thus enabling species to form viable populations. This concept is based on the theory of metapopulations (Levins, 1970; Hanski & Gilpin, 1997; Verboom et al., 2001).
For the evaluation within the project INTERMEUSE the model LARCH (Landscape Analysis and Rules for the Configuration of Habitat; Foppen & Reijnen, 1998; Chardon et al. 2000; Groot Bruinderink et al., 2003) was adapted and used for the ecological impact assessment of the proposed flood protection strategies in the Meuse catchment. LARCH is designed as an expert system, used for scenario analysis and policy evaluation. The model requires a habitat map and ecological standards or rules (e.g. on dispersal distance, population density etc.). Of each proposed flood protection strategy a resulting habitat map was predicted based on landscape ecological units. LARCH standards are based on literature, empirical studies and simulations with a dynamic population model.

The results of the habitat network analysis indicate potentials for the development of viable populations of species on the basis of the spatial habitat configuration analysed. Key elements in this approach are:

- characteristics of a species: e.g. habitat preference, home range, dispersal capacity;
- the amount, shape and area of habitat patches in a landscape;
- connectivity of the landscape, which defines how easily species can move to other habitat patches. For example, roads can seriously hamper the connectivity between closely orientated habitat patches.

With the developed method the network function of a flood protection strategy can be tested on the basis of a set of so-called ecological profiles. Each ecological profile represents a range of species with similar habitat requirements (defined in terms of ecotopes) and dispersal capacity, that can occur in a landscape. The ecological profile "Corncrake" (Crex crex) for example, stands for species that find their habitat in large patches of herbaceous grassland and have a dispersal capacity on a(n inter)national scale level. For this study, a set of 8 ecological profiles was selected (Table 5.7). For these species the current habitat configuration in the Meuse catchment area and the situations resulting from the defined flood protection strategies were analysed on the potential sustainability of viable populations. Since the assessment is based on potentials for a habitat network of a species, actual species distribution or abundance data are not required.

Table 5.7 Summary of results of the ecological network analysis for the three defined flood protection strategies (Retention, Sponge, Winterbed), compared to the present situation. o: no change; -: decrease; --: strong decrease; +: increase; ++: increase almost everywhere; (+): localised increase.

Ecotope	Ecological profile	Retention	Sponge	Winterbed
Grassland and rough growth	Large marsh grasshopper	(+)	o	++
	Whinchat	o	o	++
	Corncrake	(+)	o	++
Marshland	Bittern	+	o	o
	Bluethroat	(+)	o	++
	Large marsh grasshopper	(+)	o	++
Forest	Medium sized forest bird	(+)	o	o
	Otter	(+)	+	+
Side channels, open water	Otter	(+)	+	+

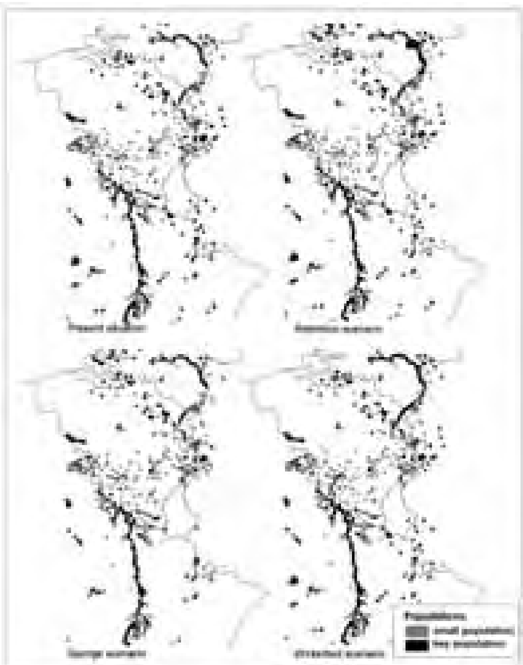

Figure 5.21 Example of results of population viability analyses for the present situation and the three defined flood protection strategies for the ecological profile "Large Marshgrasshopper" (Stetophyma grossum; Van der Sluis et al., 2001). Mvp: minimum viable population; core: key population; small: local population, too small to be a key population.

Results

The results of the habitat network analysis with the model LARCH are summarised in Table 5.6. For a complete presentation of the results is referred to Van der Sluis et al. (2001). An example of the output of the habitat network analysis performed is presented in Figure 5.21. Consequences of the spatial configuration of habitat resulting from the three flood protection strategies are shown for the potential population of the Large Marshgrasshopper.
Improvement of the network function of a landscape can be obtained by enlarging existing habitat patches or the creation of new habitat patches. Depending on type, size and shape these new patches can function as key area, stepping stone or corridor. The main objective with respect to a cohesive, viable ecological network should be prevention of further fragmentation and creation of natural areas as great in size as possible. For the Large Marshgrasshopper this would mean that a floodplain lowering strategy to maintain flood protection would result in far the most attractive spatial arrangement (Figure 5.21).
The ecological minimum was described as a critical level of habitat availability corresponding with the lowest acceptable level of ecosystem functioning (Karr, 1999). Translated to the habitat network analysis performed, the ecological minimum indicates the minimal habitat integrity for the development of sustainable populations. This can be linked to the spatial cohesion in ecological network analysis, by the potentials and boundary conditions for key populations, as minimum condition for population persistence of specific target species (Verboom et al., 2001). In the LARCH-methodology a key population is a relatively large, local population in a network, which is persistent under the condition of one immigrant per generation. In Table 5.7 indications are listed for key area size (to support a key population) and total area needed for an ecological network supporting viable populations of species. These indications are based on autecological knowledge of large numbers of species, concerning habitat demands, area needs and dispersal capacity in search of new habitats to colonise (Vos et al. 2001).

Table 5.7 Indications for the area ratio needed for sustainable networks, with and without a key area, according to Vos et al. (2001).

Species group	Key area	Sustainable network with a key area	Sustainable network without a key area
Large birds	1	4	6
Medium birds	1	3	5
Small birds and mammals	1	1.5	2
Reptiles	1	2.5	2.5
Amphibians and butterflies*	-	-	20 habitat spots

*For amphibians and butterflies not the size of the habitats but the number of habitat patches seems to be the determining factor with respect to habitat configuration.

Ecological integrity of river banks: carabid beetles

Method

As stated carabid beetles were chosen as indicator group to assess the ecological integrity of river banks. River banks are characterised by dynamic habitats and as such direct links to flooding characteristics exist (i.e. morphodynamics, water level fluctuation and flood frequencies). Based on cluster analysis of field survey data, correlations between species communities and environmental features were made. Combined with habitat requirements of indicator species a predictive model was designed, with which future situations resulting from e.g. flood protection measures can be assessed on their potentials for the integration of river bed rehabilitation goals (Jochems & Van Looy, 2001).

For the analysis of the carabid beetle communities data were collected on carabid fauna, vegetation and abiotic river bank characteristics in three pilot stretches (20km stretch each) in the three participating countries (i.e. near Mouzay (F), within Common Meuse and the Sand Meuse). In this river basin sampling 4,881 carabid beetles were counted. In a more intensive local level analysis some 80 plots were sampled in the Common Meuse stretch for two consecutive years, resulting in the catch of some 16,000 carabid beetles.

The environmental variables in the analysis were selected to have maximum ecological relevance, while being possibly influenced by flood protection measures

(Table 5.8). To identify the explanatory values of these environmental variables, a correspondence analysis (CCA) was used for the regional and local scale analysis. Based on this clustering and correspondence analysis between species communities and environmental features habitat templates are defined. These templates can be assigned to three zones within the river bank that represent the transversal gradient of river dynamics, based on the templates characteristics.

Table 5.8 River variables of channel morphology, hydrology and river bank habitats used in the cluster analysis.

Variable	Description	Measurement
River kilometre	Distance from river source (km)	regional / local
Width/depth–ratio	Dividing river width by mean river depth	regional / local
Base flow index	Dividing lowest flow by mean flow	regional
Coefficient of Variation	Dividing discharge variation by mean discharge	regional
Peak frequency (of summer discharge peaks)	Number of relevant summer peaks per summer season	regional
Peak Velocity	Hourly or daily maximum flow increment	local
Rising Speed	Velocity of water level rise	local
Habitat diversity	Number of riverbank types per station	regional / local
Texture	D50-value of substrate (mm)	regional / local
Vegetation cover	Percentage of soil covered by plants (%)	regional / local

As final step in the analysis a multiple logistic regression was executed for the explanatory river management variables. From this a response and optimum range of the variables for the ecological integrity of river banks was derived. Linkage of the defined templates to these river management related parameters resulted in a response model that can be used for the prediction of potentials for carabid beetle community development resulting from river management activities.

Based on the habitat templates and the transversal gradient they represent in the river bank, the formulated general definition of the ecological minimum is translated to a minimum available habitat within each gradient zone to allow sustainable populations of one of the communities. So, a minimum of 3 communities, divided over the 3 zones of the defined gradient, is necessary to achieve basic ecological integrity of the river bank for this species group.

Results

In the regional scale correspondence analysis (CCA) the main explanatory variables were width/depth-ratio of the riverbed and peak discharge frequency (summer season). Minor explanatory value is in habitat diversity and substrate texture. On the local level, further correlations were detected for the variables peak velocity (with the first axis 82%), and to a lesser extent rising speed of the water level (for the fourth axis 81%). Width/depth – ratio of the riverbed showed a high correlation with the habitat templates related to higher altitudes in th river bank gradient (i.e. higher vegetated bar and higher open gravel bar), which are inversely correlated with rising speed of the water level. These explanatory variables were used in the multiple regression, to build a response model for the carabid communities. Especially for width/depth-ratio of the riverbed, peak velocity, peak discharge frequency and habitat diversity, optimal ranges and responses to impacts in the system, caused by the proposed flood protection strategies, were defined, resulting in a useful evaluation tool.
The regional analysis showed that the stated ecological minimum habitat integrity was achieved in 50% of the sites monitored. The sites attaining the ecological minimum had an average species richness of 23 carabid beetles species, compared to an average of 14 for the sites with lower habitat diversity and an overall mean of 18 for the total sampling. The total cumulative species richness over the habitat diversity classes is presented in Figure 5.22. The position of the ecological minimum (as minimum habitat diversity measure) is high on the flattening curve.

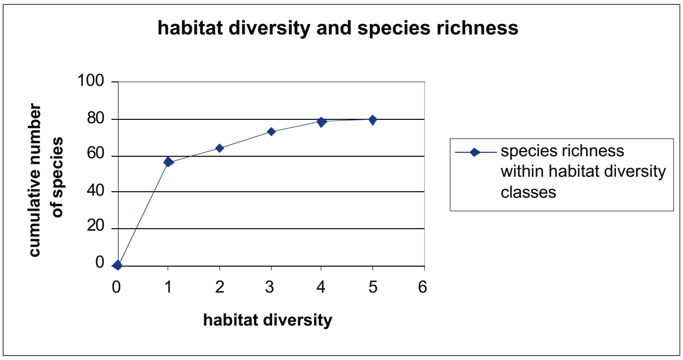

Figure 5.22 The species richness in carabid beetles over the sampling plots, cumulative over the habitat diversity range

The natural baseline (maximum habitat integrity) is achieved when all characteristic communities have sufficient habitat for the development of sustainable populations. Based on this the ecological goal was determined by interpretation of the landscape ecological unit mapping and the carabid beetle sampling results (Table 5.9). To reach the ecological minimum in one of the proposed flood protection strategies, at least three habitats should have an area corresponding to the ecological goal defined. This evaluation method is demonstrated for the WINTERBED-strategy in the different pilot stretches (Table 5.9 and Figure 5.23), based on hydraulic modelling results.

Table 5.9 Goals for rehabilitation of river bed habitats based on carabid beetle communities, with actual performance for the pilot stretches indicated as percentage of the formulated goal.

		Mouzay		Common Meuse		Sand Meuse	
Habitat	**Landscape ecological unit**	**goal**	**present**	**goal**	**present**	**goal**	**present**
Pioneer gravel bar	gravel bar	20ha	100%	100ha	10%	10ha	20%
High open bar	sandy bank	10ha	100%	150ha	10%	30ha	10%
Pioneer sand bar	sand bar	5ha	20%	25ha	5%	20ha	0%
High vegetated bar	wet border	5ha	10%	70ha	20%	15ha	30%
Wooded bar	softwood fringe	5ha	10%	50ha	5%	50ha	10%
Cut off bank	steep bank groin	2ha	80%	3ha	40%	2ha	50%
Steep bank	steep bank	1ha	100%	10ha	100%	10ha	100%
Overbank levee bar	sand bar dune	5ha	5%	20ha	15%	150 ha	10%
Flood channel	flood channel	20ha	20%	120ha	10%	400 ha	20%

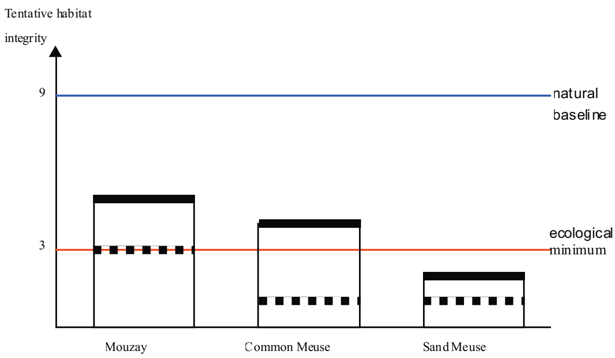

Figure 5.23 Tentative habitat integrity in the present situation (dashed line) and the WINTERBED flood protection strategy (solid line), for carabid beetle communities.

For the implementation of these ecological goals some guidelines can be stated. Principle elements in river bank habitat integrity are the river dynamics and its gradient over the river bank. A good measure for improvement of river dynamics proves to be the width/depth ratio of a river stretch. Within INTERMEUSE for each pilot stretch the variation in these parameters was assessed. The results are listed in Table 5.10 and form additional information for the ecological rehabilitation of the river bed and the integration with flood protection activities.

Table 5.10 Guidelines for river class types for the planning predictor variable width/depth ratio of the river bed (W/d-ratio).

Size/ character class	Meuse stretch	Sinuosity	Bank full discharge (m^3/s)	Ecological minimum W/d-ratio	Natural baseline W/d ratio
Upper middle course	Lorraine Meuse	>1.5	100-150 (<500)	10	30-50
Upper straight course	Ardennes Meuse	<1.5	250-500 (>100)	10	20-30
Lower middle course	Common Meuse	>1.2	1500 (>500)	20	50-100
Lower course	Sand Meuse	<1.2	1600 (>500)	18	>100

Ecological integrity of floodplains: meadow vegetations

Method

For the winter bed, meadow-vegetation communities are used as indicator group, in the same way as carabid beetles have been used for the river bank. Differences in plant species composition and zonation in floodplains can be largely explained by two major environmental factors: hydrological regime (mainly flood duration) and agricultural practices (Grévilliot et al., 1999; Grévilliot & Muller, 2002).
Based on cluster analysis data correlations between species communities and environmental features were made. For the regional analysis vegetation monitoring results from 80 rélevés from France, 60 rélevés from Belgium and 20 rélevés from the Dutch part of the Meuse were combined. The effects of interactions between hydrology and agricultural practices on vegetation spatial distribution were investigated by using a model based on CCA (Canonical Correspondence Analysis). The

CCA identifies the most important variables in predicting the probability of occurrence of the different units of vegetation. In a final step again logistic multiple regression was used in combination with GIS (Geographical Information System) to develop a predictive model that can be used for the prediction of potentials for meadow vegetation community development resulting from river management activities.

The local analysis of the impact of the proposed flood protection strategies was performed on the same pilot stretches as used for the carabid beetle analysis. The developed vegetation response model was adjusted in the Mouzay pilot stretch, as this is the most natural stretch remaining in the Meuse basin. This model is applied in the other pilot stretches (Common Meuse and Sand Meuse) as well as for the proposed flood protection strategies. A complete description of these activities within the project INTERMEUSE is presented in Krebs (2001).

As stated, the main aspects with regard to the diversity of floodplain meadow communities are the hydrological gradient (mainly flooding duration) and agricultural practices. So the ecological minimum, as minimum acceptable state of floodplain integrity that allows development and persistence of sustainable meadow communities, is based on these two aspects. The elaboration of this ecological minimum is performed for the unregulated French pilot stretch. Cluster analysis for this pilot stretch resulted in 13 distinguished vegetation groups, that in turn were clustered in four classes of meadow communities. These classes correspond to the whole hydrological gradient in the floodplain. Analogue to the carabid beetle communities, the ecological minimum was defined as a minimum of 1 vegetation group per community class (= gradient zone). Thus, a total of 4 vegetation groups representing the whole hydrological gradient should be the lowest acceptable level of ecosystem integrity based on this species group. The natural baseline is achieved if all vegetation groups are present in the floodplains. Based on the natural French pilot stretch, the ecological minimum was quantified by defining a minimum area for each community necessary to allow its persistence (Table 5.11). The connectivity with the fluvial system is an important factor for the preservation of the two wettest communities (mesohygrophilous and hygrophilous). So, spatial fragmentation in small patches of these two habitats severely hampers sustainable communities.

Table 5.11 Quantification of the ecological minimum for the different meadow vegetation communities to allow preservation. Indications are derived from the near-natural pilot stretch Mouzay (F).

Meadow vegetation communities	% of area
Hygrophilic communities	2.5
Mesohygrophilic communities	10
Mesophilic communities	5
Mesoxerophilic communities	2.5

Results

Correlation and regression analyses between the identified vegetation clusters and the determining environmental factors resulted in probability assessments for the vegetation communities. With this, for each vegetation type a vegetation response map was calculated, showing the probability of occurrence of each type. These probability maps were combined to produce a new vegetation map, based on the vegetation type with the highest probability of occurrence. In Table 5.11 the results of this exercise are listed for the Mouzy pilot stretch. With this approach potentials for meadow vegetation developments can be assessed for any given (future) situation. But, to what extent these potentials can be achieved is not only depending on the new hydrological conditions. The soil seed bank may prove to be a very important factor in this respect.

Analysis showed that, compared to the rather natural French pilot stretch, the other pilot stretches not always achieved the above formulated ecological minimum in the present situation. Both the Common Meuse and the Sand Meuse attained only 50 % of this minimum: only two communities out of four are sustainable in the present day situation. The ecological goal for the pilot stretches was set by translating the situation of the French phytosociological results to the other stretches, assuming a considerably less intensive agricultural management practice (Table 5.12).

Table 5.12 Goals for rehabilitation of meadow vegetations, with actual performance for the pilot stretches indicated as percentage of the formulated goal.

	Mouzay		Common Meuse		Sand Meuse	
Vegetation type	Ecological goal	Present	Ecological goal	Present	Ecological goal	Present
Hygrophilic communities	100 ha	60 %	490 ha	12 %	495 ha	10 %
Mesohygrophilic communities	400 ha	73 %	1965 ha	4 %	1980 ha	1 %
Mesophilic communities	180 ha	100 %	880 ha	25 %	800 ha	26 %
Mesoxerophilic communities	100 ha	52 %	490 ha	100 %	495 ha	100 %
Crops	0 ha	100 %	0 ha	100 %	0 ha	100 %

In Table 5.13 the ecological minimum and natural baseline are linked to flood duration, one of the main predictors for meadow habitat integrity. This forms essential input for the planning process within integrated river management related to the winter bed.

Table 5.13 Relation between the defined ecological minimum, the natural baseline and the relevant flood duration classes as prerequisite for the desired meadow vegetation development.

	Mouzay		Common Meuse		Sand Meuse	
Flood duration	Ecological minimum	Natural baseline	Ecological minimum	Natural baseline	Ecological minimum	Natural baseline
0 = < 1 week	2 ha	10 ha	140 ha	700 ha	100 ha	500 ha
1 = 1-2 weeks	18 ha	90 ha	240 ha	1200 ha	360 ha	1800 ha
2 = 2-5 weeks	40 ha	180 ha	220 ha	1100 ha	100 ha	400 ha
3 = 5-8 weeks	75 ha	370 ha	140 ha	700 ha	125 ha	600 ha
4 = 8-12 weeks	20 ha	100 ha	200 ha	1000 ha	120 ha	600 ha
5 = 12-20 weeks	8 ha	35 ha	115 ha	500 ha	95 ha	400 ha
6 = > 20 weeks		30 ha		400 ha		600 ha

Remark : Class 6 includes river bed and side channels

Integration of flood protection and river rehabilitation

Integrated river management implies that the new flood protection practices should at the same time focus on prevention of further deterioration of natural features and preferably lead to rehabilitation of lost natural elements. Within the project Intermeuse this was analysed for two distinct scale levels: the whole river basin and for specific pilot stretches, by using the three defined flood protection strategies. For both scale levels results of the analysis show that flood protection measures can be beneficial for nature rehabilitation aspects as well. This is elaborated in conceptual approaches and practical guidelines that can be implemented in integrated river management and spatial planning. For this, the identification and quantification of the ecological minimum for the several aspects presented before (habitat network functioning, carabid beetles and meadow vegetation) is an important step. By definition it is meant as the lowest acceptable ecological state and as such it marks the lower boundary were integration of river rehabilitation goals and flood protection can be achieved, whereas the natural baseline forms the upper boundary.

Integration on a regional scale

The performed habitat network analyses on the regional level in this project show that for the development of viable populations of species depending on typically river-bound habitats, the WINTERBED-strategy has the most obvious positive effects, especially in the Upper Meuse and in the Lower Meuse. However, since there are little possibilities to change the small winter bed in the Ardennes Meuse, this stretch appears to be a natural bottleneck for the migration and dispersal of species. Regulation of the river will however enhance this situation. The aim should be the creation of small areas wherever possible in this stretch. These can function as stepping-stones within the habitat network.
In general it might be presumed that, on the basis of the concept of ecological networks, ecological rehabilitation of river ecosystems should focus on enlargement of habitat prior to optimising habitat connectivity. For many species, one substantial area is better than a number of tiny spots (amongst others due to the larger effect of interference with surroundings, disturbance etc.). Application of the formulated guidelines requires knowledge on the present situation and formulated ecological rehabilitation targets for a river ecosystem. These targets can be based on existing nature values that need to be preserved or enhanced, or on the degree to which natural processes are still operative or can be made operative in the process of rehabilitation. Most important processes are hydro- and morphodynamics, as these are the driving forces for habitat development and diversity. These processes embody the characteristics of a certain river(stretch). This emphasises the statement that the distinguished scale levels, each having their own value within the river management process, are strongly interrelated. The influence of dynamic river processes is the most distinct on the local scale level.

Integration on a local scale

Gradients play an important role in the degree in which dynamic river processes still can influence the river landscape. Meadow vegetation and carabid beetles appear to be valuable indicators for habitat integrity for respectively winter bed and river banks. As such, a direct link is available with the type and dimensions of possible flood protection measures and river management. However, as river bed and winter bed are separate parts within the hydrodynamical gradient, conclusions on the impact of certain flood protection measures can differ in the way that measures favouring riverbed conditions will have impact on the winterbed conditions (i.e. flood duration and frequency). The decision making following the evaluation remains a balancing process, that needs to be support with knowledge and practical tools and guidelines.
With respect to the river bank, analysis of the flood protection strategies used in INTERMEUSE leads to the following guidelines. SPONGE measures can best be situated adjacent to the actual river bed. Even in small upstream parts of tributaries modified bank structures can already improve the water retention capacity considerably. Implementation of SPONGE at these sites also has a positive effect on the development of natural bank forms and the desired habitat integrity. Secondly, SPONGE measures may have a positive effect downstream: peak velocities nowadays exceed the natural conditions. Yet, a too strong decrease in peak fluxes would have a negative effect on the morphological processes necessary for habitat integrity.
Depending on the type of RETENTION measures the same recommendations as made for SPONGE are valid: the inclusion of river banks in the measures can result in an increase of habitat integrity. The effects of peak discharge reduction should be focussed on the highest and lowest peaks. In these ranges the distortion of natural flow regime is the most pronounced. The peak frequency of the intermediate range of peak fluxes is responsible for the morphological processes and hence for the development of the characteristic river bank habitats. The location of retention measures should take into account valuable floodplain areas.
WINTERBED measures should be planned in an integrated way: the combination of bed widening, bank lowering and flood channel restoration, restores the dynamic gradient in the river bank zone and is therefore beneficial for the over-

all habitat integrity. The choice for only one of the measures (e.g. bank lowering) will have effect in only one of the river bank gradient zones and as such is only partly in line with the proposed interpretation of the ecological minimum.
For the habitat integrity of the winter bed the same guidelines as stated above are applicable to a large extent. However, based on the meadow vegetation analyses another general remark needs to be made. The integration of flood protection and river rehabilitation is a good approach in strongly regulated river stretches. As this is the case in large parts of NW-European rivers this integration can lead to multi-beneficial solutions in river management. However, in near-natural river stretches any change in abiotic conditions resulting from a flood protection measure can lead to serious negative impacts on existing natural values. This brings up the question of how to combine flood protection strategies and quality preservation of natural ecosystems? In the near-natural river stretches focus is on nature preservation and less on rehabilitation. Based on the analyses for the near-natural Mouzay pilot stretch, flood protection measures should be promoted preferably in the more degraded areas as rehabilitation of lost values after implementation may never result in the natural baseline which is available now.

Conclusions

- Integration of flood protection goals and river rehabilitation goals can well be established. In regulated river systems flood protection measures can have a positive effect on achieving river rehabilitation goals. In natural river stretches combinations may be less favourable as nature preservation may be a major goal. Flood protection strategies SPONGE and RETENTION in such areas will lead to significant changes in local hydro-dynamics, which could entail important habitat and biodiversity impoverishment. Therefore, thorough studies related to the impact of management measures on habitat and biodiversity have to be carried out before implementing such strategies in natural river stretches.

- Flood protection strategy SPONGE and RETENTION should be implemented as much as possible in the upstream reaches of a river basin, as to reduce the flood peak discharges. WINTERBED-measures, that increase dis charge capacity, are the most effective on a local basis.

- On a regional level river rehabilitation should focus on enlargement of habitats and the creation of cohesive networks of habitats. On a local level the focus should be on the habitat diversity linked to gradients in the river system.

- Development of viable populations of species depending on typical river-bound habitats is served the best with the WINTERBED-strategy, in our case especially in the Upper Meuse and in the Lower Meuse. The SPONGE-strategy especially improves the situation for wetland species. The RETENTION-strategy might improve the situation for marshland species with large home range (e.g. Bittern). Considerable areas of habitat are developed under this flood protection strategy.

- Based on the habitat network analysis, the Ardennes Meuse seems to be a natural bottleneck, due the physical characteristics of this river stretch. However, river regulation will have enhanced this situation. With the creation of stepping stones this situation can strongly be improved.

- The correspondence analysis and response analysis lead to the identification of three important variables with respect to prediction of river bank habitat integrity: peak velocity, peak frequency (summer season) and width/depth-ratio of the river bed. These variables can be linked to the flood protection strategies defined in this study: the SPONGE-strategy has the strongest influence on the lowering of peak velocity; the RETENTION-strategy reduces peak frequency, and the WINTERBED-strategy influences width/depth-ratios. Responses to these variables can be predicted for flood protection measures, the resulting impact on habitat integrity can be described with the multiple logistic regression results.

- In the current situation the Dutch meadow vegetations are poorly developed and intensively used by agriculture. Restoration of the hydro logical gradient would result in an increase in moist and wet meadows. This implies a change in land use and consequently an increase of meadow biodiversity. However, the restoration of meadow vegetations in such heavily regulated river stretches might be hampered by the lack of an effective soil seed bank. This was not studied in the project INTERMEUSE.

- Win-win situations for flood protection and floodplain rehabilitation are theoretically possible. In practice the involved costs may pose the major problem for actual implementation. The concept of the ecological minimum, however, presents an instrument to quantify externalities related to flood protection measures.

- The identification and quantification of the ecological minimum is an important new guideline that may prove to be very useful in the practical integration of flood protection and river rehabilitation goals. Together with the natural baseline it defines the range where integration is possible. It should be clear however, that the ecological minimum is not meant as the general ecological goal to be achieved in integrated river management. This might only be the case in heavily modified river stretches, where due to human pressures the opportunities for river ecosystem rehabilitation are limited.

Acknowledgements

The INTERMEUSE project was part of the umbrella project IRMA/SPONGE which was managed by the Netherlands Centre for River studies (NCR) and financed by IRMA as part of the EC's INTERREG-IIC initiative. The authors wish to thank S. van den Akker, G. de Blust, J-F. Mony, F. Grevilliot, Prof. Dr. A. Musy and Prof. Ir. E. van Beek and all other persons and institutes that helped to bring the project to a good end.

VI Synthesis and conclusions

VI.1

Defining conservation objectives in river restoration: the River Disorder Approach

Kris Van Looy & Patrick Meire, in Prep.

Abstract

The manifestation of the river system is the result of an array of discontinuous, non-equilibrium processes operating at different scales, influenced by the constellation of geographic, hydro- and bio-ecoregions in the river basin. We propose a multidimensional and multiscale approach to define conservation objectives for river ecosystems. The River Disorder Approach provides a framework for deriving objectives from observed patterns and structures in the river system, resulting from the discontinuous processes among the various temporal and spatial scales. We identified disorder elements for the River Meuse at the different scale levels for the floodplain meadows and immediately derived conservation objectives from it. These where then integrated in a guiding image, to prove the practicability of this approach.
Contrasting with the common view of rivers as continuous and self-repeating in components and patterns, we identified the non-equilibrium and stochastic processes as guiding for the definition of conservation objectives. This choice conflicts with presently used deterministic approaches. As this type of deterministic approaches is used for generalized goal setting for rivers in national or even pan-European legislative frameworks, but encounters strong problems, our plead for idiosyncratic, non-deterministic target setting might prove helpful for the implementation of river basin management.

Introduction

Most large rivers and streams of the temperate regions have been drastically altered by human activity over the past centuries (Décamps et al. 1988; Petts 1989; Ward & Stanford 1995). Regulation for transportation, water supply, flood control, agriculture and power generation purposes, is recognised to come at great cost: large flood disasters at the end of last century, the loss of important natural resources and prospecting climatic changes in river catchments raised the awareness of the need for new approaches. Flood protection and harmonisation of functions need new perspectives and frameworks for the future (Giller 2005). Preservation of natural resources and restoration of ecosystem functions and health are essential elements in the development of strategies and the definition of objectives (Fairweather 1999; Karr 1999).

Whatever the initial drive to start a river restoration project: species conservation, habitat restoration, flood protection, gravel or sand extraction, water purification, there are always many parties involved to reach the point of decision and action. For the river system's hydrology, geomorphology and ecology are intimately linked, all functions and land use practices depend upon specific configurations and conditions of the river's functioning. Therefore, the development of one function can hamper many others or can be tailored towards a benefactor for other functions. Many examples exist of projects where functions are developed in such a way that they support others, or projects acquire new objectives through the planning process as win-win situations can come to light.

At the European scale, the Habitats Directive demands a clear definition of objectives for a favourable conservation status, for the species and habitats in the NATURA2000 pan-European network of protected areas, including many riverine habitats and species. Conservation objectives must represent a contribution to this achievement of favourable conservation status, and the wider goal of biodiversity conservation, for the present habitats and species based on the features for which it has been selected (EC 1992). Same counts for the Water Framework Directive that tries to initiate and organise the new perspective of integrated river basin management. It states a general objective for all water bodies, in the achievement of a good ecological status by 2015. This good ecological status achievement is subject of an integrated approach for assessing quality and goals for physicochemical, biotic and hydromorphic conditions with a common implementation strategy over the member states. It demands the definition of explicit objectives in the context of management plans and restoration strategies. Objective definition is further subject of legislations in different member states, as well as in other continents (Naiman & Bilby 1997; Boon 2000).

A clear definition of objectives in an early stage is essential and leads to the best realisation practices. The objectives for river restoration need to be realistic in relation to the natural physical processes, and their variation in time, and to the needs and demands society has brought about, and which in most instances are irreversible. Fundamental elements for the implementation of river basin management are quantitative analyses, dealing with risks, institutional organisation and the paradox of scale (Naiman et al.1998). Many handbooks and blueprint approaches for

river restoration exist (European Centre for River Restoration ECRR, Nijland & Cals 2001; River Restoration Centre 2005; National Rivers Restoration Science Synthesis 2004; River Styles Framework, Brierly & Fryirs 2002), yet, most start from the idea of a universal river character. We present a stepwise approach for the definition of conservation objectives, starting from a disorder concept for rivers. It is an approach based on the discontinuities and heterogeneity in the river system, not starting from unifying principles.
The conservation objectives have to be made explicit within the context of biodiversity conservation, and then translated in decision frameworks. Quantitative measures have to be derived and integrated in restoration schemes. From our multidisciplinary research on the River Meuse, in light of the large-scale restoration project for the Common Meuse reach, emerged the here described conceptual framework to develop and prioritize restoration strategies.
For the introduced approach, two central questions are:
1 how do we define the objectives for biodiversity conservation and restoration
2 how can these be measured? Quantitative, tangible measures need to be defined for objectives

We first refer to existing approaches and methods in the definition of objectives, then introducing the River Disorder Approach and its application to the river Meuse. At the end we discuss the concept's merits and the gaps and constraints in existing frameworks and legislations for successful conservation and restoration of river systems.

A. Existing approaches

Reference conditions

As suggested in the legal frameworks, conservation objectives are in the first place derived from reference conditions. Reference conditions may be based either on historical or geographical comparisons or on modelling, or may be derived using a combination of these methods including historical data. When no references are at hand, conceptual frameworks are consulted to derive model or indicator approaches.

The few studies that have documented attributes of relatively intact or notionally pristine rivers (e.g. Ward et al. 1999a; Radwell & Kwak 2005), and countless studies that have provided detailed reconstructions of river evolution over timescales of decades, centuries, or longer (Petts 1989; Girel et al. 1997; Décamps et al. 1988), indicate just how profound human-induced changes to river forms and processes have been across most of the planet. The European Water Framework (WFD, EC, 2000) nevertheless does demand the definition of reference conditions, if not of a historic or actual reference, than derived from a retracing of impacts to communities (Wallin et al. 2003).
Biological conservation and restoration strategies often refer to 1900 as a reference situation for Western European cultural landscape before industrialisation and land use intensification (Haslam 1996). Proposed restoration measures, classified as mitigation by Boon (1992) concern piecemeal land use practices and internal management of hydrologic and soil conditions. The ecological integrity goal or natural baseline (Karr 1999, Jungwirth et al. 2000) for these strategies is determined for particular communities and/or species under specific management regimes of mowing or grazing. River restoration in the temperate region refers more often to a reference situation further back around 1800 as the larger river regulation works started around that time (Figure 6.1). And even this situation deviates from the unaltered pristine conditions (100% integrity) before the large landcover changes in the catchments took place.

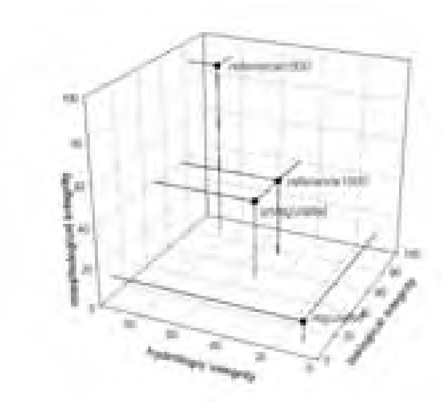

Figure 6.1 Reference conditions and restoration pathways in terms of biological, morphological and hydrologic integrity.

For the rivers of our temperate regions, most river alterations were already largely present in 1900. The deterioration of biological integrity since, is mainly due to further flow regulation and/or to intensification of land use. Further hydrological deterioration is caused by embankment and gravel or sand extraction, resulting in bed incision and distraction of large floodplain area. The unregulated reaches can readily be seen as reference for the regulated reaches, as they offer interesting prospective emphasizing on the definition of reference conditions and targets in the context of the WFD. Aquatic communities might even recover to a level comparable with the less disturbed unregulated reaches, even through immediate influx of species (Usseglio-Polatera et al. 2002). These river reaches are, however, only comparable to a certain degree, for some conditions and/or taxonomic groups (Pedroli et al. 2002).

Conceptual approaches

In river ecology, the most important conceptual framework for biodiversity patterns in the river system is the River Continuum Concept (Vannote et al. 1980). It depicts a gradually changing biotic community in equilibrium with the physical environment of river systems from headwater to mouth, as physical, chemical and biological processes vary with river size. The concept states the important differences in ecological processes such as energy flow, organic matter breakdown and community structure in river channels along a longitudinal continuum.

In this way it follows the logic of Strahler's river order (Strahler 1957). Since the definition of the RCC, many contradictory observations and fundamental criticisms were the starting point for the definition of new concepts. From observations of strong discontinuities in geomorphologic and hydrologic regime, Statzner & Higler (1985) came to the formulation of the Stream Hydraulics Concept. Further concepts that make the variance and dynamics in hydrologic regime tangible for objectives are the Range of Variability Approach (Richter et al. 1996) and the Natural Flow Regime concept (Poff et al. 1997), concentrating on river specific flow variation and disturbance regimes. The discontinuities of both natural and anthropogenic origin in the system, can generate a regular pattern in processes and community structure, as is depicted in the Serial Discontinuity Concept (Ward & Stanford 1995) and the Telescoping Ecosystem

Model (Fisher et al. 1998). A further addition to the RCC based on local discontinuities due to strong lateral exchanges in large floodplain rivers, is the Flood Pulse Concept (Junk et al. 1989). This concept emphasizes more on the merits of the river dynamics and especially flooding processes. These dynamics and the forthcoming disturbance patterns are also the scope of the Patch Dynamics and Shifting Mosaics concepts (Petts & Bradley 1997; Forman 1995), interesting frameworks in the light of habitat network and population strategy approaches. Both concepts define equilibrium conditions with the physically changing environment over time and space in the river corridor, subject to disturbances and dissipation of energy, but not in a continuous or orderly form.

Functional versus structural approaches

Approaches to define objectives can be functional or structural. Productivity and nutrient cycling oriented approaches offer solutions to many management and quality related objectives. In these functional approaches the ecosystem health is in the first place defined in goals for nutrient cycling, buffer capacity and resilience, integrating discharge energy and water quality goals. Structural biodiversity approaches start from a well-defined appraisal of biological integrity and biological endpoints. In rivers, the physical structure of habitat is defined largely by the movement of water and sediment within the channel and between the channel and the floodplain. While reduction of environmental heterogeneity reduces options for species diversity (Naveh & Lieberman 1994), the ecological heterogeneity in river systems is closely related to flow regime and flood pulse characteristics, influenced by river management and floodplain land use. So, in both kind of approaches, objectives have to entail an array of factors.

Further we have to stress the scale-sensitivity of objectives. Different spatial scales require different target setting and actions as for example conditions of riparian corridors differ at reach or basin level.
Differences in the magnitudes and rates of many of these factors are governed by differences in discharge, channel width, channel depth and other management-related features. Scale-sensitive approaches to rivers are proposed moreover, determining functional units to the river system and the watershed management (Sear 1996). Different approaches apply to different river scales of basin, reach or site

(Stanley & Boulton 2000). In this way, we can present some materials for objective definition in this perspective of functional and structural approaches (Figure 6.2).

At the catchment scale the functional elements have much more weight in the definition of objectives than at the local scale where approaches are directed immediately at tangible structural conservation objectives.

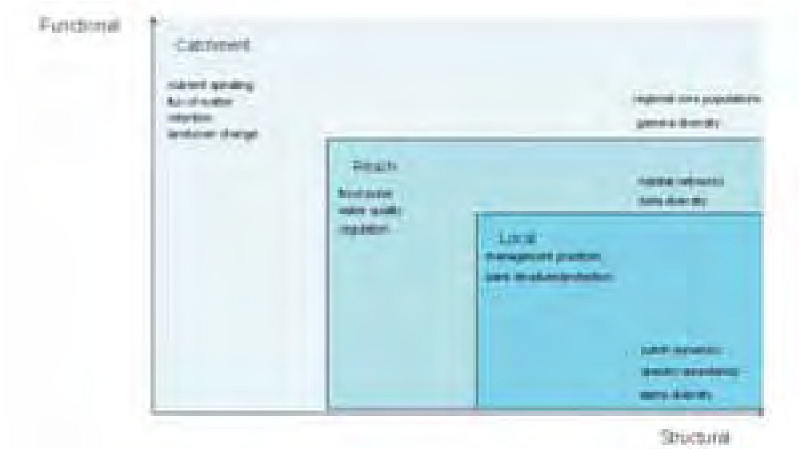

Figure 6.2 Presentation of functional and structural descriptive measures for objective definition in the watershed context.

So, conservation objectives can be defined in many ways; they can take the form of normative descriptions, qualifying conditions, units of target species or habitat. From the introduced definitions and frameworks we can derive 4 target fields for conservation and restoration, in which we can look for useful measures:

1. River corridor reservoir: the biodiversity of the fluvial system in terms of species populations and gene pools. Reviews on environmental interactions in riverine communities and river corridor species (Malanson 1993; Naiman & Décamps 1997; Naiman et al. 1999).
2. Connectivity: reviewed by Malanson 1993, Forman 1995, Gregory & Petts 1996.
3. Natural flow regime: reviews on this field and its interaction with conservation are given by Poff et al. 1997 and Growns & Growns 2001.
4. Morphodynamic equilibrium: reviews of this component by Leopold

(1994); Steiger et al. (2005) review the hydrogeomorphic processes of unconfined alluvial channel-floodplain rivers within the temperate zone, and Hughes ea (2001) its relation to riparian biodiversity.

Within these target fields of the river ecosystem conservation and restoration context, we can screen what kind of measures and approaches exist in objective definition (Table 6.1).

Table 6.1 Published data on conservation approaches within the four target fields for rivers/catchments:

Target field	Approach	Quantified objective parameter/measure	Data dimension	Reference
Reservoir	Watershed analysis of habitat objectives	Fish habitat in Large woody debris, pool frequency, stream temperature, aquatic insects	Siuslaw River, USA	Kershner 1997
	River health concept, with multimetric Index of Biotic Integrity	Bentic invertebrates and fish taxa, diversity	Kissimmee River, USA	Karr 1999
	Biological modelling Salmon populations	Water temperature	Grande Ronde basin, USA	Watanabe et al 2005
	Functional-geographical approach	Riverine pasture community patches	River Dinkel reach, NL	Wolfert et al. 2002
Connectivity	Riparian corridor function	Seed input Northern and	River networks, Garonne rivers	Nilsson ea 1989, Tabacchi ea 1996
	Incidence function metapopulation model	Spatial population dynamics	Drainage basin, USA	Lowe 2002
	Network Dynamics hypothesis	Channel networks structuring riverine habitats/communities	River basins, USA	Benda ea 2004
	Patch dynamics concept and habitat templet theory	Habitat templets /patches	Small rivers, GB	Townsend ea 1997
	Hydrologic connectivity	Material and organism transport	Reach Donau, Austria	Ward ea 1999b Piégay 2000
	River habitat networks approach Econet	Species habitat networks, dispersal capacities	River Rhine, NL	Reijnen ea 2001
Flow regime	Natural flow regime	Magnitude and frequency, timing, duration and rate of change	River systems	Poff et al 1999

Target field	Approach	Quantified objective parameter/measure	Data dimension	Reference
Geomor-phology	Range of Variability Approach	Peak magnitude and timing	River reaches, USA	Richter ea 1997
	Instream Flow Incremental Methodology	Depth, velocity and substrate varying with discharge	River reaches, USA	Bovee & Milhous 1978
	Streamflow model	Flood duration	Colorado River reach, USA	Auble ea 1994, Shafroth ea 2002
	Physical Habitat Simulation System (Phabsim)	Physical variables	Rivers	Bovée 1982
	Habitat and species assemblages	Bank profile and structure	Reach	Armitage et al 2001
	Hydrogeomorphic (HGM) method	Functional Capacity Index for physical/ biotic variables	watershed	Whigham et al 2003
	River Styles	Geomorphic features, framework	Rivers/catchments channel form	Brierley & Fryirs 2000
	Measures of physical diversity	Thalweg, cross-section, variability of sediment size	Reach, Creightons Creek, Australia	Bartley & Rutherfurd 2005
	Reversibility and readjustment of channelized rivers	Specific streampower	Smaller rivers of Denmark and Great-Britain	Brookes 1988
	Stream stabilization	Sediment supply	catchment	Sear 1996
	Erodible Corridor Concept	Dynamic river corridor	Reach, Ain, Marne: France, Po: Italy	Piégay ea 2005
	Riverine Ecosystem Synthesis	Functional Process Zones	River networks	Thorp ea 2006

Where do problems arise and do most approaches fail in the definition of objectives?

Ideally the definition of biodiversity conservation objectives should include information on a variety of different taxa and be carried out at different scales and in different landscape ecological units, as biodiversity patterns are scale-sensitive (Wiens 1989). Nevertheless for many river projects objectives are formulated monospecific (e.g. for Salmonids), mostly leading to unsatisfactory results, as measures are ineffective or conflicting to other formulated objectives (Frissell and Nawa 1992). The pressure of timeframes, tangible results, and political interests has lead to a preponderance of short-term, transitory rehabili-

tation projects that ignore the underlying capacities and developmental histories of the systems under consideration, and seldom place the treatment reach in its catchment context (Ebersole et al. 1997, Lake 2001, Bernhardt et al. 2005). Restoration efforts typically have been directed at the site level, yet suffered from a lack of ecological understanding of watershed processes at the ecosystem level and have sometimes done more harm than good (Frissell et al. 1993; Doppelt et al. 1993).
Besides these failures caused by restricted investment in dimension and scale of the projects, some general points of concern for most approaches can be raised. Firstly, most approaches aim at deriving overall solutions and generally applicable principles. Especially in the context of regional typologies and legislative contexts, objectives are defined and solutions proposed as more generally applicable, for groups of rivers rather than river-specific.
The complexity of river functioning and the heterogeneous nature of stream and riparian conditions enlarges the risk of failure as specific conditions demand specific solutions and cumulative or threshold effects can occur. Cumulative effects of restoration practices arise when impacts accumulate and generate unwanted effects. Threshold effects refer to the responses of biological elements to restoration activities, which are often nonlinear relationships. For this reason, geographical or historical references do not offer target images and measures that can simply be transferred to actual conditions of a site to be restored. The scale-sensitivity, complexity and idiosyncrasy of the river system's functioning and processes, hampers these generalizing approaches.

Second critical aspect arises in the translation of the approaches towards objective definition for the biotic system. Mostly specific species groups or single target species are focussed in this exercise of deriving quantitative measures and objectives. The resulting measures often comprise species abundances and numbers, habitat suitability indices, species groups metric indices that are integrated in objectives for habitats. Best examples are conservation plans for fish species or species groups' spawning or breeding sites. Discussion between species-based and processes-based approaches is not new, but with respect to the complexity of the river system and its community patterns and processes, the process-based approaches offer the best perspective to our opinion.

The River Disorder Approach

Referring to the conceptual approaches, Statzner & Higler (1985) already pointed at a first aspect of disorder in the river system with the hydrologic discontinuities in the river resulting in changes in aquatic communities. We observed discontinuities in floodplain meadow communities along the river in relation to adjacent ecoregions and the configuration of different ecoregions in the catchment (Van Looy et al. In press).

These observations lie at the basis of the definition of the River Disorder Approach, that we present here as a scale-sensitive approach to the definition of conservation objectives. The River Disorder Approach points at the ability of the river system to adopt the variability of geology, landform and climatic conditions in the catchment to its appearance and identity over its course, expressed in discontinuous patterns along longitudinal and lateral dimensions and in ecological patterns of diversity and structure in its biological and physical component. It is not just another aspect of disturbance in the river system we point at, but more the integration of influences of disturbance/perturbation/landscape origin operating in the river system, leading to the characteristic heterogeneity in the river system (Table 6.2).

Table 6.2 Discontinuities and heterogeneity can be determined in the fields of hydrology, geomorphology, biogeography and biotic processes.

Disorder field	Natural disorder element	Anthropogenic disorder element
Hydrology	Confluence	Weirs
	Extreme peak event	Water abstraction to canals
	Tidal impact	Hydropeaking
	Peak velocity	
Geomorphology	Geologic discontinuity	Gravel/sand extractions
	Slope	Normalisation/regulation
	Bank (in)stability	Embankment
Biotic reservoir	Ephemeral habitats	Species eradication
	Disturbance strategies	Introductions
	Stochastic assembly	
Biogeography /	Ecoregion contact	River corridor fragmentation
Connectivity	River corridor discontinuity	Isolation of floodplain area
	Extinction-colonization dynamics	Creation of new migration pathways (canals)

The River Disorder Approach points at a crucial aspect of river ecology in the light of defining objectives for conservation and restoration; notably the natural dynamics and discontinuities in space and time as sources of heterogeneity at the different scale levels, present in geomorphic, hydrologic, geographic and biologic context. Several authors documented on the unpredictable character of the river heterogeneity, at different scales (Pollock et al. 1998) or under different regional settings (Sabo et al. 2005), and the relevance for conservation and restoration options.

The river system is an ever changing environment driven by different scale disturbances or disorder elements and a biotic system that shows a strong complexity in local ecological patterns across various temporal and spatial scales. Figure 6.3 presents these river disorder elements in their spatial and temporal scale of impact.

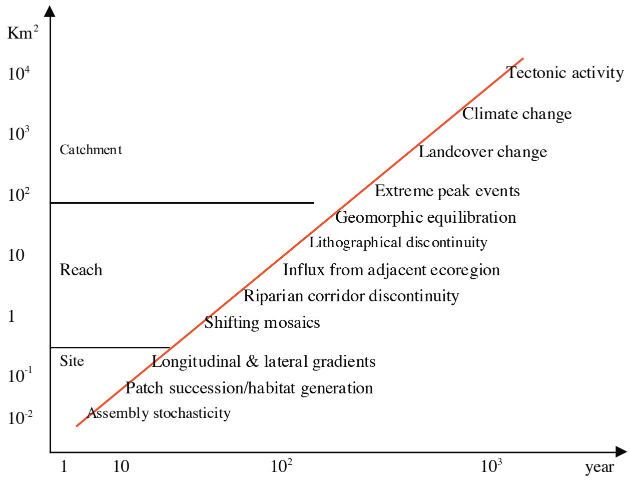

Figure 6.3 The river disorder elements in their spatial and temporal scales.

Salo (1990) presented a similar graph arranging in this spatiotemporal scale the fluvial geomorphic processes and their tentative biotic correlatives. He stressed the equilibrium responses of successions, life history strategies and biological differentiation, yet also gave some ideas of instability and discontinuous responses, like channel migration, extreme floods and fluvial dynamics creating patches varying from <1m? to tens of thousands of square kilometres. Shugart (1990) in the same book stressed non-equilibrium responses as crucial factor in river systems. He

referred to discontinuities generated in nature in non-equilibrium systems.
The disorder elements at regional scale are often proposed as classification elements for river systems. The most widespread are (hydro)ecoregion delineations, integrating data of hydrogeology together with valley forms (Wasson 1992; Petit 2002). Reaches determining elements are defined in geomorphic and geographic characteristics of substrate, valley form and slope (Frissel et al.1986; Rosgen 1994; Sear 1996; Ebersole et al. 1997; Myers & Swanson 1997; Cohen et al. 1998). These typologies are useful in explaining changes in riverine communities. Schumm (2005) goes further in this reach scale analysis and reviews 36 elements destabilizing the geomorphic equilibrium of particular reaches as causes for incision for river beds. Within these causes, he included not only perturbations and disturbances to the river bed conditions but also natural fluctuations and even biotic processes impacting the physical system. So, in fact he determined an array of disorder elements of geologic, geomorphic, climatic, hydrologic, biotic and anthropogenic origin.
We identified some additional disorder elements at different scale levels (Figure 6.4); regional climatic differences over subcatchments, the connectivity and patch dynamics of the river corridor and the contact with the uplands and adjacent ecoregions causing discontinuities in the biotic system,.
In Figure 6.4, the cumulating of disorder elements in the river basin is illustrated, from the different geographical and regional climatic character of subcatchments, to the impact of the biogeographic regions and landscape configuration in contact/connection with the river system, and on the landscape and site level, illustrated for discontinuities in patch mosaics, habitat heterogeneity and community assembly.
The connectivity which is one of the main characteristics of the river corridor and the river-floodplain system, can show discontinuities and disorder in space and in time, due to anthropogenic as well as natural disturbances/alterations. The shifting mosaics of the river system (Naiman et al. 1988) can be seen as the result of two contrasting tendencies, towards homogeneity and heterogeneity, according to Pinay et al. (1990). He describes discontinuities of different origin in the ecotone conceptual approach; discontinuities at confluences as result of the complexity of the hydrographic network, or changes due to alterations of slope, reflected in changes or mixing of communities. He also depicted the non-equilibrium aspects of the river system in this ecotone approach.

Same counts for Thorp et al. (2006) who describe riverine ecosystems as nested, discontinuous hierarchies of patch mosaics, dominated by non-equilibrial and stochastic processes. These authors see these processes responsible for the formation of a quasi-equilibrial, metastable state of rivers, portrayed as downstream arrays of large hydrogeomorphic patches (e.g. constricted, braided and floodplain channel areas) formed by catchment geomorphology and climate. So, with this notion of quasi-equilibrium, they try to derive rules/generalizations for the patch mosaics.
The disorder concept depicts the riverine ecosystem as a complex, discontinuous system displaying structures that reflect the influence of the river basin constellation of georegions, hydroregions and bioregions and the processes determining fluxes of matter and species.

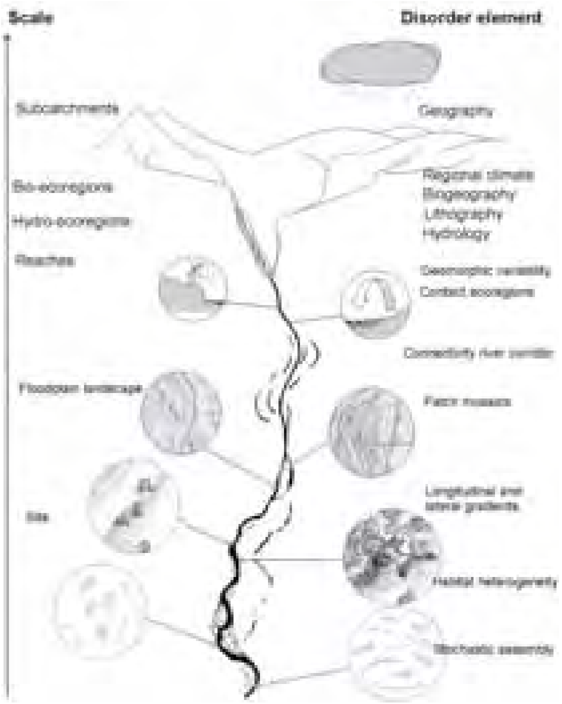

Figure 6.4 Disorder elements in the river basin. The inset figures illustrate the discontinuity at the different scale levels caused by the specific disorder elements. The map insets are taken from single maps of reaches, with same legends for exactly adjacent sectors/stretches.

These disorder elements can be tailored towards the definition of conservation objectives at the three scale-levels of the river system, here illustrated for the River Meuse.

A multiscale approach

The River Disorder Approach looks for distinguishing features at different scale levels, explaining habitat heterogeneity and biotic diversity of the river system. Key factors contributing to the river's disorder character are flow regime related disturbances and gradients, geomorphic variety and morphodynamics, connectivity in longitudinal sense but also laterally with adjacent landscapes and ecoregions, and biotic reservoirs in their specific relation with the environment. In contrast to the commonly used species-based approaches to the definition of conservation objectives, it is a processes oriented approach.

In a first step (Table 6.3) the river basin is screened for sources of variety, rather than searching unifying elements as is done in most conceptual river frameworks - looking for the unifying principles in and between rivers. The disorder is detected between reaches, with the distinction of geomorphic, hydrological and biogeographic entities at catchment level. Differences in communities and gamma diversity can be attributed to disturbance regime, changes in environmental conditions and influx from adjacent ecoregions.

Table 6.3 Description of steps in the River Disorder Approach.

Step 1. River basin level	Step 2. Reach level	Step 3. Local level
Discontinuities in geomorphology delineate	Beta-diversity analysis determines steering processes reaches	Alpha diversity of patches and species environment relationships are determined
Gamma-diversity and dissimilarity analysis reveals key factors	Longitudinal and lateral gradients and heterogeneity results	System processes and management practices in habitat conditions
Driving variables are derived from geomorphic, biogeographic or anthropogenic origin.	Gradients can be of natural as well as anthropogenic origin.	Biologic integrity for species groups/communities with emphasis on stochastic and non-equilibrium conditions and compositions.

The River Meuse, one of the larger rivers of the European Western Plains ecoregion, can be divided into 6 reaches based on geomorphic and hydrological characteristics (Figure 6.5). The two free-flowing reaches of the Lorraine Meuse and Common Meuse both have wide alluvial plains, whereas the other reaches are all regulated narrow floodplain-river systems.
The river's rain-fed character with torrent peak flows and a flow rate ranging from 30 to 3000 m^3/s, causes the riparian corridor to be highly impacted by the unpredictable hydrologic regime and catastrophic events.

Discontinuities and disorder in the catchment were revealed in the composition of floodplain meadow vegetation. In its middle course a high complexity of ecoregions and tributary confluences is present. In figure 6.4, the arrows for ecoregion contact indicate the influxes of species from distinct ecoregions in the catchment, as was observed in our data (Figure 6.6). The high disorder in the middle reaches of the river was determined in physical as well as biotic aspects of diversity (Figure 6.7, 6.8). The dissimilarities (Figure 6.9) in the floodplain meadow communities between the reaches shows high resemblance between the outer reaches I and VI, whereas for reaches I and III the highest overall dissimilarity with other reaches is present.
From the analysis of disorder in terms of geomorphic and hydrologic changes, ecoregion input and important biotic reservoirs, objectives and guidelines for conservation and restoration approaches can evolve. For reaches (like the Common Meuse) with high disorder, emphasis in conservation must be on promoting natural disturbance processes and influx of species from the surroundings. For the low disorder reaches of the Lorraine and Tidal Meuse, floodplain rehabilitation can be designed more isolated from the surroundings or the upstream/downstream influences.

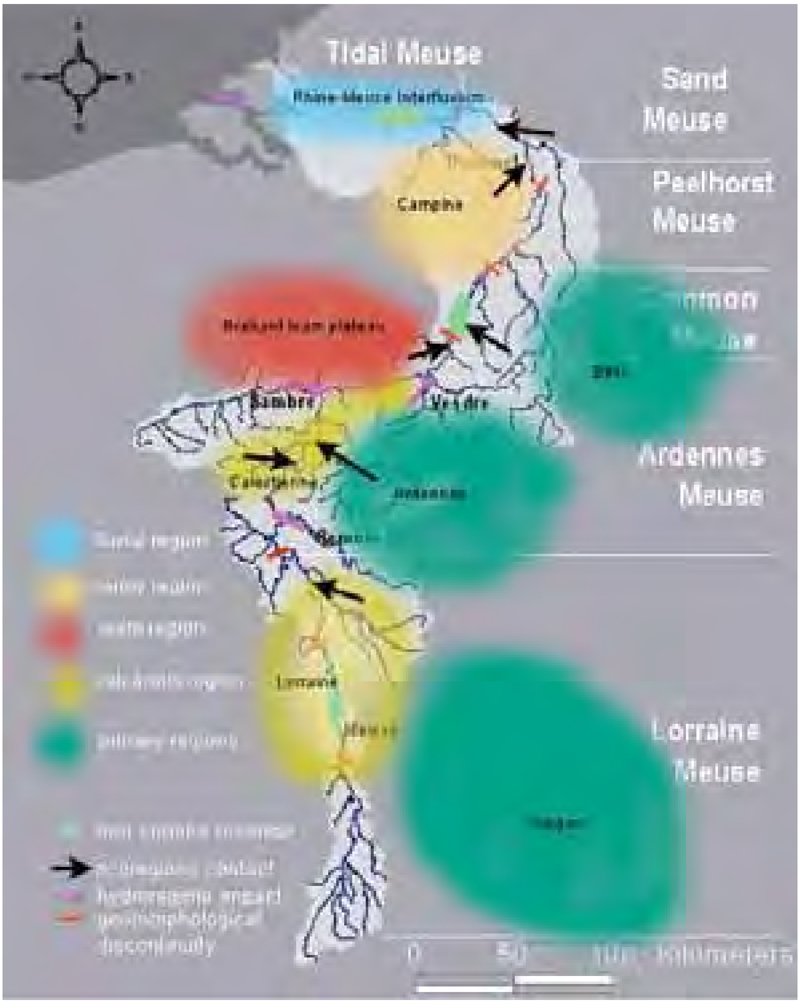

Figure 6.5 Map of the River Meuse basin with indication for the main disorder contributions at catchment's scale of the 4 components of disorder: flow regime, geomorphology, biotic reservoirs en connectivity.

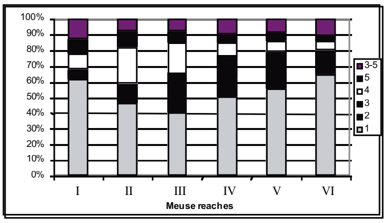

Figure 6.6 Species ecoregion partitions of the plots of the Meuse reaches (1: fluvial region, 2: sandy region,3: loam region, 4: calcareous region, 5: primary region

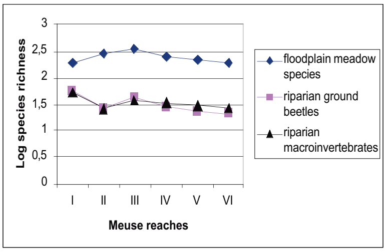

Figure 6.7 Species richness in Meuse surveys of different groups over the reaches. For floodplain vegetation, the species richness peaks in the middle reach, for the aquatic and semi-aquatic organisms, the unregulated upstream reach shows the highest species richness, still there is also in these groups a strong recovery in species richness in the middle reach.

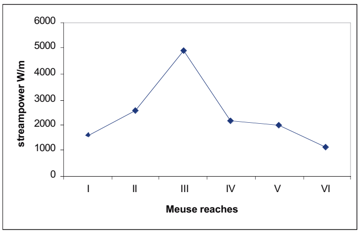

Figure 6.8 The stream power in the different Meuse reaches.

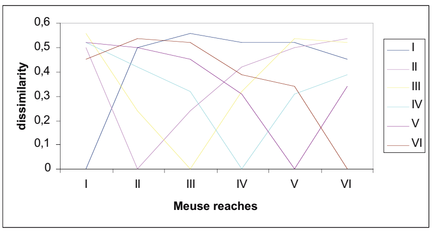

Figure 6.9 Coenocline dissimilarity projections of qualitative similarity along the river.

After the screening at basin level, and the delineation of reaches, in a second step the diversity within reaches is focussed. The reach scale disorder is governed by the stochasticity of flood events with strong shifts of energy, material and populations in time and place. The disorder in hydroregime and morphodynamic conditions is reflected in composition and diversity of communities in the river system in lateral and longitudinal gradients and patch mosaics.
These aspects were determined for the floodplain meadows in the population dynamics strategies (following Freckleton & Watkinson 2002) of the species at risk in these communities, the rare river corridor plant species (rare = less than 5 populations in the study area). Emphasis was on the dry river grassland communities, the main protected habitat in the NATURA2000 network for this area. Strong lateral gradients were documented for the floodplain meadows over the Common Meuse alluvial plain. Disorder was present in isolation caused by riverbed incision and dike construction that disconnect parts of the alluvial plain from river influence. The population dynamic strategies for the rare river corridor species are mostly remnant, patchy and metapopulation strategists (Table 6.4); typical strategies for species at risk. The colonisation index and extinction risks per population strategy group show the disorder elements at this scale level. The disorder is most tangible in the groups of remnant and metapopulation strategies, showing low colonization, linked to the disconnection and isolation from the river flooding. The colonization index differs strongly from more stable population strategies, indicating the determining role of seed dispersal in plant metapopulations undergoing recurrent local extinctions and colonization (Tilman 1985). Further it is important to distinguish populations with low extinction probabilities from populations with high extinction probabilities in the light of conservation and restoration options.
We revealed the necessity of dynamics for the conservation of these species, as we observed that dynamic habitats, with species in more dynamic strategies, show highest potential to recruitment and restoration in general. Furthermore, with respect to the catchment scale analysis, we observed influx from adjacent ecoregions to be highest for reaches with high disorder character (highest variability in hydrologic and morphologic conditions), mainly for the use of various population dynamic strategies.

Table 6.4 Colonization rate and extinction risk of rare river corridor plants within the population dynamic strategies of Freckleton & Watkinson (2002).

Strategy Remnant	Patchy	Meta-population	Source sink	Shifting cloud	
share of species (%)	24	55	16	2	4
colonization index	1.31	4.1	1.38	4	6
extinction risk	10	28	42	50	70

Colonization index is an observed recruitment rate (group average of colonised patches/species/peak) for the rare river corridor plant species within the Common Meuse floodplain survey. Extinction risk is the probability of disappearance, measured in the percentage of highly dynamic sites for a species (averaged for each population strategy group).

Floods and hydrodynamics are not only responsible for a lateral gradient in community composition, they were also significant parameters in determining habitat generation and succession. In this way, they are a discontinuity and disorder element, as the flood peak events prove to be a crucial element in generating habitat and in seed dispersal. So, the community composition and diversity is for its spatial and temporal pattern more explained by the infrequent large floods and only to a lesser extent by the regular flooding.

Table 6.5 The River Disorder Approach results derived from the floodplain meadow analysis of the River Meuse.

Scale	Disorder element	Driver	Measure
Catchment	Discontinuity in community composition	Ecoregion influx and river dynamics	Stream power and adjacent ecoregions contact
Reach	Discontinuity in patch mosaics, in riparian corridor connectivity and in species strategies	Infrequent large floods	Flood power and floodplain gradient/ alterations
Site	Stochastic assembly and site idiosyncrasy	Extreme local heterogeneity and unique conditions in space and time	Habitat heterogeneity and species diversity

In step 3, this analysis is taken up to the level of patches' alpha diversity and local habitat conditions. At this level, the disorder can be determined for specific communities or species groups. The most important factor sustaining high biodiversity at the local level, is habitat heterogeneity (Rosenzweig 1995). This was revealed in the diversity analysis of the floodplain meadows emphasizing on the presence of rare river corridor species. The exceptional high slope of the species-area relationship (highest species richness corresponds to small patch size) shows the effects of habitat heterogeneity and fragmentation of habitat in the Common Meuse floodplain. This is illustrated for the individual patch richness in Figure 6.10 with the indication of the relicts rich in rare species. The smallest patches are the richest in rare species, and even generally most species-rich. This illustrates the stochastic character of community assembly in the riverine landscape. No stable structure or dependence on local environmental conditions for communities were observed, no pattern of saturation or equal distribution for patches in different states was present. We can conclude that the specific context of disturbances and landscape features is responsible for the erratic/stochastic distribution patterns of habitat patches and species, and thus for the non-equilibrium conditions for communities in the riparian corridor.
As we observed riverine communities to be loosely structured and not saturation-oriented, and as larger patches in river systems tend to be poorer in species diversity, maximizing heterogeneity is a good option for biodiversity conservation in the river system.

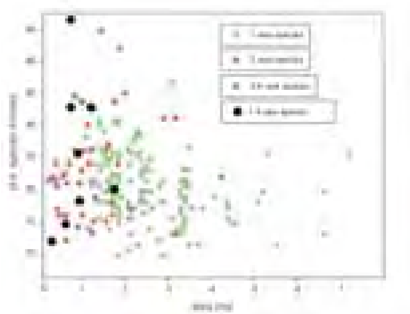

Figure 6.10 The species richness-area relationship for the individual plots of the Common Meuse floodplain meadows. The graph shows that the smallest patches are the richest in rare species, and even generally most species-rich. This is a proof of the extreme habitat heterogeneity in the river system.

Bringing about this concept; a multidimensional approach in generating conservation objectives

These observed disorder elements, determined for the most relevant physical and biotic elements, can generate tangible measures and guide conservation and restoration options.

Figure 6.11 Observed disorder elements for the River Meuse at the different scale levels, along the functional and structural axis

Figure 6.11 presents the measures we quantified for the River Meuse at the different scale levels, based on study of floodplain meadow vegetation (see above), riparian forest (Van Looy et al. 2003, Van Looy et al. 2005a) and riparian ground beetles (Van Looy et al. 2005b).

With this analysis, the River Disorder Approach can be seen as a multiscale and multidimensional approach for defining conservation objectives and prioritizing river restoration strategies. The early establishment of a 'guiding image' with a dynamical ecological end state is seen as the most critical aspect of river restoration projects (Giller 2005). This guiding image must specify how (i) the system works, (ii) it has been impaired (i.e. the key stressors and how they impact on ecosystem health), and (iii) the intended restoration alleviates or reverses the key stressors (Jansson ea 2005). The defined conservation objectives have to be translated in comprehensive forms for decision frameworks and managerial plans. A set of key ecological factors at reach and local level identified in the River Disorder Approach, with tangible measures and model application, can be used to make a guiding image of the restoration project. For the Common Meuse, the following factors were selected, based on identified responses in specific communities: flood frequency, peak velocity, flooding power, habitat fragmentation and the need for

sediment supply (eroding banks). The relationships between composition and diversity of these groups and the physical variables were quantified with general linear regressions and thresholds were determined for the disorder elements. In this way quantitative measures were generated.
The guiding image (Figure 6.12) shows for a Common Meuse stretch all the criteria for restoration in a way that is comprehensible for river managers and other possible stakeholders.

Figure 6.12 Guiding image for the Common Meuse restoration project. It shows the different disorder components starting from the geomorphic near-equilibrium conditions in the historic situation, over the biotic reservoir in habitat and species relicts, the connectivity in flood channels and riparian forest corridor on to the flooding regime in sedimentation zones. All these key factors were integrated in the restoration objectives, defined in preservation of relicts, connection of natural areas in the riparian corridor, provision of eroding banks and lowering of banks to allow flooding and the development of riparian forest.

5. Discussion

5.1 Approach

Stochastic processes and non-equilibrium conditions in both the physical and biotic compartment of the river system are at the basis of the disorder concept. Shugart (2005) describes the scale-aspects of disturbance and landscape with respect to equilibrium conditions as follows; quasi-equilibrium landscapes are much larger than the disturbances that drive them, and show a relatively constant proportion of patches in a given successional state. Landscapes influenced by a disturbance regime whose spatial scale extent is so large that it could be termed a catastrophe, or landscapes whose dynamics and proportion of patches in differing states are subject to chance variation, are non-equilibrium landscapes. In this respect, he documented the large rivers as effective non-equilibrium landscapes, based on the relation between spatial extent of floodplain forests and the spatial extent of floods (Shugart 2005). Wiens (1984) proposed for biotic communities a gradient from equilibrium to non-equilibrium in the following characteristics: saturation, competition, stable or loose structure and stochasticity of disturbances. Looking at the community level, we observed several causes for disorder and non-equilibrium in composition and diversity of communities. Where saturation and the striving for equilibrium are the classical foundations for community ecology, we observed mostly unstable and non-saturated assemblies. Most determining for community composition proved recruitment and dispersal limitation, extinction by fluvial or anthropogenic disturbance and responses to the changing physical conditions by resilience or disturbance strategies.
Fitter et al. (1999) point at temporal variations leading to disequilibria at a point in space and to the possibility of coexistence of species which could not coexist if competition was allowed to proceed. We think most species assemblies in river systems can be categorized as non-equilibrium communities. Strong indications were documented for: 1) aquatic macroinvertebrate assemblies, for which strong disorder is observed in the Meuse in frequent consecutive invasions of exotic species last decades (bij de Vaate et al. 2002); 2) riparian ground beetle assemblies responding to extreme local heterogeneity in flow dynamics (Van Looy et al. 2005b); 3) floodplain meadow vegetation for which the species-area relationship and the need for stochastic processes/extreme flood dynamics observed for the

rare river corridor species, proves the opposing trend to saturation. So, these non-equilibrium communities are loosely structured assemblies with species more responding to environmental variations largely independent of one another (Wiens 1984). Especially the major contribution of stochastic events (extreme peak flows) to species dispersal and colonization/extinction, proves determining for observed diversity and composition patterns. As the persistence of small populations is strongly affected by stochastic problems (Foose et al. 1995), our analysis of the population dynamic strategies of the rare river corridor species highlighted the threats for the characteristic river species of the Meuse. Many riverine species only persist as metapopulations in the fragmented habitats and it may therefore be insufficient to protect small areas along a river to save its riparian communities (Andersen & Hanssen 2005). Conservation efforts should neither be oriented in preserving and maintaining local conditions and communities, as these are part of the ever-changing riverine landscape. Objectives should be tailored towards the non-equilibrium conditions and the driving forces behind them. In this way, we think that this disorder approach highlights crucial aspects of riverine communities and provide useful frameworks for the definition of conservation objectives for river restoration and conservation programmes.

5.2 Application

Two key concepts drive the River Disorder Approach: the non-equilibrium and the idiosyncrasy of the river and its (biotic) components at the different scale levels. This implies that no deterministic approach can be followed to derive objectives, and that for each river/reach/site specific objectives are needed. To prove the merits of the River Disorder Approach, we discuss the outcome of classical approaches to the River Meuse's natural resources (Table 6.6).

Generalized (nationally derived) objectives for protected habitats or river-types
Conservation objectives derived for the protected habitats and species for the Common Meuse reach, the 50km border reach between Flanders and the Netherlands, result in mitigation measures trying to stop further alterations in physical conditions and deterioration by habitat loss. With the dry river grasslands as main protected habitat, no restoration efforts to the riverbanks and

the contact between river and floodplain would be envisaged, as these might alter present assumed-critical habitat conditions. So, here a conflict might arise.

On the Dutch side of the same Common Meuse stretch, a large river restoration programme was initiated (Van Leussen et al. 2000). This project starts from a specific reference situation as target (more conform to the WFD objective definition). A uniform set of measures will be reproduced all over the reach; the riverbed will be widened and banks lowered over the whole river reach.. The following aspects of our River Disorder Approach, presented in the guiding image, are not envisaged with this project approach: 1) Local relicts of dry river grasslands are not regarded, 2) eroding banks to supply the river with sediment for the restoration of morphological processes are not integrated, 3) parts of the floodplain will be disconnected from regular flooding and 4) riparian forest restoration will not be allowed as this is seen as a threat for the flood protection objectives of the project.
Some crucial elements and measures of our approach, as presented in the guiding image, are overlooked or underrated in both the 'classical' conservation and restoration approaches.

Species-based approaches to objective definition

For the protected habitat of the dry river grasslands and the threatened river corridor species, several authors argument the necessity of protecting existing populations outside the river's influence (Jongman 1992; Hegland et al. 2001; Vervuren et al. 2003; Eck et al., 2005) and reintroducing species and habitat in an artificial way (Stroh et al. 2005). Yet, we found evidence for their need for river contact and river dynamics. We did not determine the dispersal limitation but the recruitment limitation to be the major threat, as we observed a strong ability to colonize newly generated patches (Van Looy & Meire in Prep).
The same conclusion counts for the protected fish species, present in disconnected river arms. Here is discussion to the reconnection of this habitat to the river, as contact with the river might favour predator species. Yet, their extinction probability is extremely high in this isolated habitat. So, with regard to the natural disorder character of the habitat, we argue that a natural flood contact with the river might favour the population survival in the long run.

Table 6.6 Comparison of approaches and measures in the classical approaches and the River Disorder Approach to conservation and restoration of river systems.

Classical approaches	River Disorder Approach
APPROACHES	
• equilibrium oriented	• non-equilibrium oriented
• uniform, general rules	• idiosyncrasy oriented
• deterministic approach	• freedom for stochastic character
Strict conservation approach	*Disorder conservation approach*
• disturbance mitigation	• dynamics rehabilitation
• species and habitat preservation	• dynamic community/habitat approach
• habitat restoration	• process restoration
Classic restoration approach	*Disorder restoration approach*
• reference/'Leitbild' oriented	• disorder features oriented
• continuity oriented	• discontinuity oriented
• single species- or habitat-based approach	• complexity based approach
• one-dimensional approach	• multidimensional approach
MEASURES	
• relict preservation, e.g. isolated patches	• restoring contacts
• preserving species, e.g. restocking fish	• preserving populations in their spatial and temporal context
• managing up to community level, e.g. hayfield restoration	• free community assembly, result of dynamic conditions in space and time
• preserving actual habitat conditions, e.g. fish in stagnant cut-off branches	• rehabilitating dynamics, maximizing connectivity
• uniform bank lowering	• locally preserving erodible banks
• putting back forest development for reasons of flow resistance	• leaving it up to the river to maintain its flow section
• restricting hydropeaking by limiting hydropower production	• remediating hydropeaks by bed widening that dampens peak velocity

Possible win-win situations with the River Disorder Approach immediately come to light in Table 6.6. The merits of the River Disorder Approach lie in the combined effects of the multidimensionality and multiscale analysis of problems and key factors. From the level of discontinuities at river basin level onto local rules of assemblies, the identified disorder elements provided strong insights in key factors and communities to the detection of measures and the definition of objectives, as they play a major role to biodiversity in its functional and structural organisation of communities.

5.3 Gaps and constraints for effective river restoration programmes; problems in definition of objectives for non-equilibrium and river-specific conditions

It is difficult to effectively manage non-equilibrium landscapes toward a goal of constancy because they are regularly disequilibriated by disturbance events (Shugart 2005). Nevertheless this type of goals is present in the pan-European legislative contexts of the Habitats and the Water Framework Directives. Although these legislations tried to integrate a sense of dynamics; this still conforms to quasi-equilibrium conditions of constant proportions of patches and more or less stable structures of communities. Furthermore, every river, every reach has its identity, posing critical questions for designers and managers (Pedroli et al. 2002; Décamps 2005). This idiosyncrasy is hard to handle in legislative frameworks that cover different countries or entire continents.

So, the European legislative frameworks show the same risks of failure as the presented concepts and approaches in the perspective of river restoration. The comparison of applications for both a Water (WFD) or Habitats directive (NATURA2000) proof approach for the Common Meuse (par. 5.1), showed the shortcomings of both deterministic approaches and the merits for the proposed River Disorder Approach.

For the Meuse basin, the NATURA2000 network of protected areas under the Habitats directive comprises large parts of floodplains in the upstream reach and less in the middle course, in the lower course large parts of the estuary and the aquatic system are protected. The Habitats Directive implementation is mainly strictly preservation oriented. The protection can even come to hamper river restoration projects, as these should be oriented to a dynamic habitat approach. For the Water Framework Directive, as a result of a centralised organisation, no river-specific approach is possible and for all types of rivers the same (kind of) options and objectives will result from the pressures and impact analyses. For both legislations, generalized approaches to define conservation objectives and favourable status are applied to/over larger geographical regions (countries), leaving less freedom to analyse the habitat or species in its regional setting. Especially for river systems, this context of spatial and temporal coherence, as was described in terms of appearance and character of the river (Pedroli ea 2002), is essential and works over larger regional scales than those addressed in the habitat and species conservation approaches.

For our Meuse example, as the NATURA2000 and WFD guidance documents do not mention terrestrial riparian fauna, riparian forests, river corridor plants or sediment deposition habitat, these elements are overlooked in the present exercises for the WFD implementation and in the development of management plans. We believe the here presented River Disorder Approach for river restoration adds important elements for successful restoration and conservation attempts.

General conclusion

Goal setting for conservation and restoration efforts in river systems mostly starts from the assumed presence of equilibrium or quasi-equilibrium conditions that can guide the planning and measures. We believe that the dominance of non-equilibrial and stochastic processes in riverine landscapes demands a non-deterministic, idiosyncratic approach, that we proposed in a conceptual model for river restoration, named the River Disorder Approach.
The non-equilibrium state together with the uniqueness and idiosyncracy of riverine landscape processes, patch mosaics and community assembly within the river discontinuum, were illustrated for the River Meuse with emphasis on the floodplain meadow communities. We highlighted what knowledge of disorder elements and non-equilibrium conditions can add to develop good conservation strategies and define clear objectives at the same time.
New aspects to the existing concepts on variability and discontinuity in the river system in the River Disorder Approach are 1) the cumulating of disorder elements in spatial and temporal context, indicating that no equilibrium can be sustained and 2) some newly observed sources of variability in river ecosystems, especially in lateral relations to the river.
As we observed riverine communities to be typically in a non-equilibrium state, objectives and strategies for conservation need to be dynamic. Benchmark projects disregarding stochastic processes and site-, region- and catchment-specific potentials and constraints risk failure. The highlighting of discontinuous patch mosaics and non-equilibrium community structures also contrasts to traditional conservationist approaches. We can conclude that this identified non-deterministic, idiosyncratic character of riverine processes and communities poses problems for many generalizing approaches and legislations and demands adopted approaches to the defining and prioritizing of conservation objectives and restoration strategies.
Conclusion for strategies and policies is that a non-deterministic approach for objective formulation is needed, treating each patch, each part of a river system as a unique feature. The River Disorder Approach integrates these aspects in a multiscale, multidimensional approach.

VI.2

Conclusions

The disorder concept

Perhaps the 'furious' character of the River Meuse with the torrential peak discharges and strong changes along its course inspired the definition of a 'disorder' concept. But of course this concept is not completely novel. The river system has been described for its discontinuities in aquatic (Statzner & Higler 1985) and terrestrial riparian organisms (Ward & Stanford 1995, Tabacchi et al. 1996), the merits to biodiversity of the heterogeneity of the riparian zone associated with the fluvial dynamics is well-known (Pollock et al. 1998; Ward et al. 1999), the role of large, infrequent floods was topic for a special issue of BioScience (Michener & Haeuber 1998), and for the physical character of rivers a raising attention goes out to managing the stochasticity and comprehending the non-equilibrium conditions (Billy et al. 2001).

Where we tailored this disorder concept towards an approach to biodiversity conservation, it is nowadays also acknowledged in the search for flood protection strategies at catchment's scale. For the biodiversity conservation approaches, we concluded that a single overall strategy for biodiversity conservation of the river or its floodplains is not feasible. Strategies can, however, be derived for separate river reaches based on the reach's disorder characteristics and the influence of surrounding ecoregions.
For the flood protection strategies, the same conclusions can be drawn. To the attenuation of the highest peak flows, no general approach at basin level is effective. Only a differentiated approach and set of measures for retention and peak attenuation for different tributaries and subcatchments can prevent the peak flow cumulating over the different tributary confluences that is the cause for the extreme peak levels. For each site or reach, an array of characteristics from alluvial soils, tributary influence, floodplain width and stream flow section, together with the configuration of the upstream and downstream basin, play a determining role in the decision for appropriate flood protection measures.

For the biotic system, the disorder was described in non-equilibrium biotic communities. The dry river grasslands are a good example, as they are present in the floodplain thanks to but also despite of the river dynamics. These communities are mixtures of species typical for dry conditions all over the catchment, so for each river they show a strongly varying character in composition. Flood events are necessary to generate habitat by overbank sedimentation and for the dispersal of the plant propagules. However, these species are flood intolerant, not enduring longer inundations, wet soils nor nutrient enrichment. This illustrates the disorder character of the river system, that we appointed as crucial element for the river's biodiversity and its conservation and restoration. Others question the merits for conservation of the disorder and heterogeneity of the riparian landscape. For the vegetation in the river corridor, Mouw and Alaback (2003) questioned the contribution to biodiversity conservation for the aspects of extreme habitat heterogeneity and hyperdiversity, as the floodplain might function as sink habitat for a major part of the present species. These authors argue that only specialist species of river corridors are of conservation interest.

We described the river as an open system. To the disorder aspects contribute the entropy patterns of natural material, organism and energy dissipation and disturbances, present in physical discontinuities and connectivity aspects of contact with subcatchments and confluences of tributaries. River networks dissect landscapes and provide a natural framework for conservation planning, with distinct additive value to biodiversity conservation, if these indeed influence diversity patterns (Sabo et al. 2005). In our observed patterns for the River Meuse, we found prove for the influence of the dissected and connected landscapes in the river network. Influx from adjacent ecoregions is highest on reaches with high disorder level, mainly for the use of various population dynamic strategies, allowing for high biotic diversity. These hyper diverse sinks play a role of major importance to biodiversity conservation for populations can be viable for longer periods within the river's 'sink'. River systems have been illustrated to play a key role in larger habitat networks and the remediation of fragmentation (Wilcox and Murphy 1985; Sluis et al. 2001; Verboom et al. 2001). The link between fragmentation and biodiversity and gene flow in the river corridor (Zwick 1992; Imbert and Lefèvre 2003; Van Looy et al. 2003) points at the crucial aspect of room for riparian ecosystems and their connectivity by fluvial processes in the present situation of alluvial plains. Disturbance patterns and catchment's crossing points influence potentials and management recommendations for biodiversity conservation. The connectivity along the river as well as lateral to the river (with adjacent ecoregions) proves significant to sustain local biodiversity patterns, which react to local dynamic physical conditions. Whereas reaches with lower dynamics are more independent of upstream energy, material and propagules. The last can be treated isolated from other reaches and hinterlands, and plans can be elaborated locally-based. For instance for the Lorraine Meuse the contact with adjacent ecoregions, which proved very low in species composition, does not prevail in conservation strategies, in contrast to the conclusions of Mouw and Alaback (2003) for upstream reaches. For reaches with high disorder, the contact with adjacent reaches and regions does count.

As congruence in diversity patterns among different taxonomic groups is generally low in river systems, as the groups respond differently to major environmental gradients and/or pressures, the selection of type-specific, pressure-specific indicator groups for assessment purposes is an important element for future research and the elaboration of operational frameworks in the WFD (Heino et al. 2005). In our research, we emphasized on and highlighted responses of different communities for different aspects of the river functioning. In defining the hydromorphological reference conditions, we showed the specific status of riparian forests and riparian landscape dynamics of larger rivers. The study of riparian ground beetles revealed a group with strong indicative power to flow regime and river management. The dry river grasslands and river corridor plants revealed aspects of connectivity and patch dynamics in the floodplain, where species only persist as metapopulations in the fragmented habitats. Guidelines and targets were defined for specific local or reach scale or even catchment scale conservation strategies, based on determined responses of specific communities.

Too centralized and generalized objective definition and frameworks for developing river basin management plans and defining restoration measures, risks to pass by the unique natural heritage that evolved from thousands of years of river processes in its specific geomorphic, runoff and biogeographical setting. River-specific key ecological features need to be identified for conservation and restoration target setting. Guidelines and targets were derived for specific local or reach scale or even catchment scale conservation strategies, based on determined responses to disorder elements of specific communities. The definition/enhancement of a river-specific approach is especially necessary for the definition of restoration and conservation measures for endangered habitats, communities and species. Most rivers need specific analysis on historic and present alterations and impacts to define effective measures to reach a good ecological status, as is also documented in literature (Ebersole et al. 1997; Tockner et al. 1999; Poff et al. 1997).

The disorder approach in the River Meuse biodiversity conservation and restoration

The disorder approach offers a framework for assessing restoration potentials, defining and quantifying objectives. In the elaboration of this approach, some side steps were made to tools and instruments in river restoration programmes, like modelling and evaluation methods.

The approach distinguishes between the 4 components over the 3 scale levels; for each the analysis can be pointed at descriptors, pressure-impacts and target measures. The resulting disorder elements and measures from our study, are gathered in Table 6.7.

Table 6.7 Disorder elements and measures for descriptive phase, pressure-impact analysis and target definition over the river system components and scales.

		Catchment	Reach	Local
Reservoir	**D**	populations of river spp., core populations	community diversity, beta-diversity	habitat templet diversity, alfa-diversity
	P	viable populations river species in the river corridor	riparian forest extent, invasive species	isolated relict stands
	T	core habitat	riparian forest extent	# populations river corridor plant species
Connectivity	**D**	gamma-diversity, dissimilarities	metapopulations, riparian forest patches	isolated relicts
	P	riverine communities	young forest stages	Endikement
	T	similarity	young forest patches	patches dry river grassland
Hydroregime	**D**	peak amplitude	peak frequency	flooding frequency
	P	amplitude, variability, low flow duration	flood frequency	peak velocity
	T	range of variability	flooding gradient	peak velocity
Geomorphology	**D**	land cover/land use dynamics	Stream power, bed geometry	overbank sedimentation zones
	P	land cover change	w/d ratio, embankments	w/d ratio, sinuosity
	T	sediment load, bed	width/depth ratio transport	erosion-sedimentation rate

Step 1 Catchment analysis

The described observations at the basis of the reaches delineation give an impression of the river's identity in appearance, succession and character as depicted in chapter II.1. The most widespread used are hydroregion or ecoregion delineations at catchment scale, integrating data of hydrogeology together with valley forms (Wasson 1992; Rosgen 1994).
The different character of rivers in different parts of the catchment, results in different responses to pressures. Cohen et al. (1998) found different behaviour for water courses between regions in the Loire basin for valley slope and stream order. For alluvial rivers, slope and order explained mesohabitat distributions, in contrast with the cohesive rivers of the upstream catchment regions where no predictive power was found for mesohabitat distribution in the slope and stream order.
We found for the Meuse basin little predictive power in order for biodiversity and community composition aspects for riparian ground beetles, nor for floodplain meadow vegetations, yet strong influences for disturbance indicators of hydroregime and geomorphology between reaches. For riparian ground beetles, no regional habitat could be determined. The floodplain vegetations showed clear patterns of regional influences.

Step 1. Catchment's scale analysis

1 *Hydroregime.* Disturbances of natural as well as anthropogenic origin can generate specific biotic conditions and are of the most important river identity aspects (Pedroli et al. 2002; Angermeier & Winston 1997). The amplitude of flow variation is a widely used descriptor for hydroregime conditions and alterations (Poff et al. 1997; Richter et al. 1997). The River Meuse's flash flow character is increased by changes in land use in the catchment, yet in the lower flow ranges a bit diminished by flow regulation (damming) and for some reaches in these lower ranges enlarged by hydropower generation, with hydropeaking. The combination of land cover changes and accelerated water runoff, together with water abstractions to canals mainly, prolonged low flow periods in the Meuse with problems of water quality and scarcity. Tangible measures can be a flow amplitude

measure like Richter's Range of Variability, useful in defining the natural conditions and type of river. The Range of Variability Approach is a useful approach in defining targets for hydrologic conditions of amplitudes and flow management.

2 *Geomorphology.* Sedimentological aspects, slope and valley form determine spatial and temporal dynamics of the riverscape. Same as for hydroregime, land cover change is also one of the main causes for changes in morphodynamic conditions in the river, as it influences the sedimentological character. Quantitative measures for sediment load, bed transport or mesohabitat distribution can be defined at catchment scale based on geomorphological and geographical characteristics of substrate, valley form and slope (Cohen et al. 1998; Ebersole et al. 1997; Frissel et al.1986; Rosgen 1994; Myers & Swanson 1997). These regional approaches of hydroregions and ecoregions are nowadays used as basic descriptors in legislative and management frameworks. Petit & Pauquet (1997) proposed a typology for the rivers in the Ardennes ecoregion of the Meuse basin, based on bankfull discharges. The understanding of the geomorphology of all water courses in the network of the river basin is an important step in the development of river basin management plans (Petit 2002). The proposed regional typology allows predictions of sediment transport (especially of coarse gravel bedload) and geomorphic influences of subcatchments at basin scale.

3 The *connectivity* in the catchment is source for similarities as well as dissimilarities in communities. The latter when high influxes of species from adjacent ecoregions or subcatchments cause important changes in communities between reaches. For the Meuse floodplain meadow vegetation strong dissimilarities were observed that could be attributed to the spatial configuration of ecoregions in the catchment. Connectivity aspects of the contact and influx of species from adjacent ecoregions/ subcatchments are documented as characteristic feature for river systems determining the conservation value of landscapes at regional scale (Mouw & Alaback 2003; Sabo 2005). Gamma-diversity analysis at reach level, or over landscape compartments can show important aspects of the connectivity contribution to biodiversity. The landscape configuration determines the role of fragmentation and contact with the hinterland for the river system. Pressures can be observed in the disconnection of landscapes leading to impoverishment of riverine communities and overall diversity, due to fragmentation (observed in floodplain forests) and isolation (observed in river

corridor plants). Tangible measures for the connectivity between reaches and with the surrounding landscapes can be similarity metrics, indicating the relative importance of longitudinal and/or lateral contacts and allowing choices for reconnecting landscapes or fluvial corridors.

4 Descriptor for the *river corridor* reservoir are viable populations of river species, in which we define river species as species adapted to natural river system conditions. In contrast, exotic and invading species mostly reflect alterations and deterioration of the system. Regional key species, processes and gene pools can be defined. Distributional and human-impacted constraints for key species like Salmon and Sea trout, as well as for dominance and functional aspects of exotic invasive species in macroinvertebrate communities of the Rhine and Meuse were observed by bij de Vaate (2004). The presence of exotic invasive species can have geographical as well as functional controlling mechanisms (Tabacchi et al. 2005), as was observed for the Box elder (Acer negundo) in four gravel rivers of the Western plains ecoregion. For the same rivers, the presence of poplar (Populus nigra) and willow (Salix purpurea) functional processes and gene pools proved determining for restoration potentials. For Populus nigra this gene pool effect was well documented for aspects of genetic drift (Imbert & Lefèvre 2002) and genetic competition (Vandenbroek et al. 2004) in river systems. For - viable populations, in the light of genetic drift and competition in the river corridor, the provision of larger core habitat for key species is essential. This definition of core habitat and guidelines in habitat configuration are useful in the target setting (see II.2).

Step 2 Reach level analysis

At the reach level, processes and gradients need to be identified that are determining for community composition and diversity (beta-diversity) as well in longitudinal as in lateral sense. Longitudinal gradients can be of natural as well as anthropogenic origin. In the Common Meuse, the gradual dampening of a hydropeaking perturbation of low flow regimes, produced by hydroturbines of an upstream power plant, was observed in a longitudinal gradient in composition and diversity of the riparian ground beetle community, impact was measured up to 40 km downstream. Strong lateral gradients were documented for the forest community over the Common Meuse alluvial plain. Disruption from river influence by endikements proved significantly negative for diversity of forest communities. The disturbance origin of this compositional and diversity change brings us to the processes functioning at reach level. Floods and hydrodynamics were not only present in a lateral gradient for floodplain forest diversity, they were significant parameters in determining riparian forest development for which spatial and temporal sequences were defined and specific river forces and controlling mechanisms were found. The power and frequency of the flood pulse also determines river grassland communities of the floodplain. The soil and sediment enrichment with nutrients and alkalinity correlates significantly with flood frequency and distance to the river respectively.

1 *flow regime:* From the five critical components of the Natural Flow Regime: Magnitude, Frequency, Duration, Timing, Rate of change (Poff et al. 1997), the first was determining at catchment level. At the Meuse reach scale, peak frequency and amplitude were identified as the most important elements of the flow regime at reach level. Where the frequency is the most tangible factor over the reach, the amplitude can show gradients with the bed profile and flood plain contact. For both aspects examples of quantitative relationships with community diversity exist. Changes to flood frequency by construction of levees are important pressures for threatened communities of dry river grasslands. Flood frequency gradients and projection in space and time in a modelling approach, makes way for effective restoration measures.

2 *geomorphology:* The morphodynamic character of the reach can be quantitatively described by the bed geometry and substrate. Normalisations and embankments strongly influence especially the bed geometry and the morphodynamic character. The impact of these pressures is a shortage in gravel and sand sediment load for morphodynamic rehabilitation. Therefore the provision of eroding cut off banks was documented as necessary source for the rehabilitation at the Common Meuse scale. For the restoration planning, guidelines can be derived from the measures of potential disturbance. Quantitative values for the planning and action phase can come from constituents of the morphodynamic character of the reach, the bed geometry and substrate. The stream power is a measure for the morphodynamic activity and rehabilitation potential at reach level and with the specification to the specific stream power guidelines for bed geometry can be derived. Brookes (1988) defined a auto-rehabilitation

boundary condition (35 W/m^2) for minor regulations to small rivers, which we extended for the use in larger rivers. We found relations with the width-depth ratio for riverbank communities. The bankfull discharge can be a guiding measure for bed profile (Qbf ~ width/Depth) and sediment/substrate parameter targets in restoration projects.

3 *connectivity:* Descriptor are specific species or groups of species, that can illustrate the connectivity along the river or lateral/transversally. For amphibians the contact between floodplain and hinterland can be important for annual migrations, reported essential for the protected Triturus cristatus (Liefveld et al. 2000), yet of no meaning for others that are flood tolerant. The detection of population dynamic strategies allows the identification of important corridor functions identified for metapopulations along the River Meuse as well as for remnant populations with hinterlands. The contact and influx of species from adjacent ecoregions/hinterlands, depends on the valley form and local geographic conditions. For young riparian forest stages, the necessity of contact at reach level was signifi cantly highlighted. Elimination of young forest threatened populations and necessitated reintroduction of Populus nigra along the Common Meuse, as relict stands had become isolated. As measures for restoration, the presence of patches of young forest per river kilometre can be identified. 0,6 ha per river kilometre of the young riparian forest stage was revealed as minimum habitat requirement for a viable population of Black poplar and Purple willow along large gravel rivers.

4 *reservoir:* Descriptors were identified in habitat templets of ground beetles and beta diversity gradients in floodplain meadow vegetation communities over the reaches in longitudinal and lateral sense. These gradients and templets were related to river processes. The conservation of this diversity is documented as immediately related to river functioning, for instance for naturalness of riparian vegetation (Tabacchi et al. 1996). Viable populations of river species can depend on habitat quality or quantity conditions, as was documented for riparian vegetation extent, for forest and river corridor plant communities. Pressures are the disruption of river functioning by the construction of levees, dikes. Fragmentation of riparian habitat (e.g. for riparian forest) is a threat for diversity as well as viability of

populations. Perturbations of gradients and natural disturbance regimes increases the risk of invasions of exotic species. Measures were derived in extent of riparian forest or other habitat for viable populations of key species and integrated in modelling.

Step 3: Local scale

The local level analysis focusses at species-environment relations and alfa-diversity of patches. Most important factors at this level sustaining high biodiversity in the river system, are habitat/landscape heterogeneity and the flood pulses (Branciforti et al. 2003). The biological integrity can be determined for specific species or species groups (e.g. habitat templets), in terms of river management as was documented for Carabid beetles, minimum area of habitat for viable populations, recurrence of flood events for most groups (riparian ground beetles, riparian forests and dry river grasslands) and the specific configuration of habitat in terms of fragmentation/isolation problems for river corridor plants and floodplain forests.
The dynamics of habitats in terms of succession or flooding regime can be guiding for management options and conservation strategies. For specific species identified as key species, conservation actions or reintroductions can be appropriate, as for beaver and black poplar was initiated along the downstream Meuse reaches.

1 *flow regime*. Flooding frequency was identified as critical parameter for diversity of ground beetle communities and floodplain vegetation at the local scale, as it governs the most important habitat conditions at this scale. Natural disturbances of flooding can generate specific biotic conditions and are of the most important river identity aspects (Pedroli ea 2002, Angermeier & Winston 1997). Observed peak velocities and changes caused by hydropower generation (hydropeaking), showed linear responses with diversity of gravel bar communities over the Common Meuse reach. Local measures for the weir dampening in peak velocity were formulated to solve this problem.
2 *geomorphology*. Overbank sedimentation zones were determined critical habitat for the Natura 2000 protected areas of the Common Meuse. Sedimentation/erosion processes are measured in patch numbers of overbank sand/gravel sediment zones as habitat for dry river grassland. Embankments and the regulation activities (observed changes to width/depth ratio) resulted

in a decrease of habitat. The restoration has to envisage erosion-sedimentation rates for specific locations.

3 *connectivity:* Remnant populations of river corridor plants. Isolation of populations in the riverine landscape mosaic was observed for river grasslands caused by habitat fragmentation and disruption from river influence. Threats for these remnant species were identified or by endikement or by deteriorated stand conditions due to intensification of agricultural practices. Guidelines for contact with the river for the floodplain, or for natural disturbance regimes with regular generation of new habitats and recruitment events were formulated for the dry river grasslands of the Common Meuse reach. Metrics of minimum numbers of patches for river corridor plants.

4 *reservoir:* At the local level, alpha diversity of patches and habitat templet diversity can be evaluated . High diversity of patches was linked to the presence of rare river corridor plant species. River corridor plant species are highly threatened by isolation and fragmentation caused by intensification of agricultural use. Targets can be quantified in habitat criteria of threatened River Corridor Plants of dry river grassland , and species like Salix purpurea, Populus nigra (criteria for minimum viable populations in terms of habitat configuration/minimum patch number/area).

Guidelines from this approach for biodiversity conservation and river restoration

General concluding guidelines

In order to be sustainable, management and restoration of regulated rivers must be based on the principles of the multidimensionality, of the non-equilibrium, of the idiosyncrasy and of the dependence upon a certain level of disturbance of the river system.

For the definition of endpoints for river restoration, a few general guidelines can be derived. Where it is generally assumed that a high biodiversity and the presence of specific target species marks the endpoint of the rehabilitation,

many questions remain of how far and soon this process runs. Some little emphasized aspects of the target setting in river restoration: appearance, succession and character, together comprising the river's identity, can be guiding in the assessment of indicators, indices and tools for evaluation. The riverine landscape and its habitats show for each river a unique spatial and temporal coherence and character, a central element of the river's identity in the definition of conservation objectives and restoration measures.
The identity of the River Meuse was highlighted in a range of steering processes and critical boundaries for communities and species that were identified as river-specific. Not the species or communities were the starting point, but the relationship between the biotic and physical system was starting point for the analysis.
From the identification and quantification of key elements, an important question for the restoration is raised with the 'ecological minimum', the critical boundary or minimum level of habitat conditions for a good ecological functioning.

Figure 6.13 Spatial habitat network coherence analysis for the protected species Whinchat (Saxicola rubetra) at the Meuse basin level (Geilen et al. 2001).

Conservation objectives and legal requirements can be guiding in this analysis, which needs to be executed at different scales: a) from the catchment level (see Figure 6.13), b) over the reach level (e.g. Table 6.8), c) up to the local prediction of protected habitat types to develop after restoration (Figure 6.14).

Table 6.8 Guidelines for Meuse reaches for the Width-depth ratio (Geilen et al. 2001).

Size/character class	Meuse stretch	Sinuosity	Bank full (m^3/s)	Ecological minimum W/d	Natural baseline W/d
Upper middle course	Lorraine Meuse	>1.5	100-150 (<500)	10	30-50
Upper straight course	Ardennes Meuse	<1.5	250-500 (>100)	10	20-30
Lower middle course	Common Meuse	>1.2	1500 (>500)	20	50-100
Lower course	Sand Meuse	<1.2	1600 (>500)	18	>100

The width-depth ratio was determined as a good measure for the river dynamics and the river equilibrium. Together with the ecological status description, the equilibrium in riverbed measures (W/d) determines the resistance-resilience to extreme flood events for a river stretch. This parameter was introduced as 'elasticity' ('veerkracht' which refers to resilience) in the Dutch water management legislation (Vierde Nota Waterhuishouding 2000; Vis et al. 2001).

Disorder and biodiversity conservation strategies for the river Meuse

These observations and rules can be applied to define guidelines for biodiversity conservation strategies for the different Meuse reaches. The floodplain meadows of the upper and lower reaches with large natural floodplains showed the lowest diversity (chapter II.2). So, the naturalness and width of the floodplain in these reaches has no immediate trade-off in biodiversity. Nevertheless, the numbers and share of fluvial species are maximal in these reaches. In the upstream reaches of the Meuse the high share of species that are selective for habitat characteristics (avoiders, resisters) and the dominance of remnant and patchy population strategies shows the adaptation to the fine-scaled landscape mosaics present. The habitat management aspects with respect to the landscape pattern prevail in conservation strategies for the upstream Lorraine Meuse as was already documented (Grévilliot & Muller 2002; Vécrin et al. 2002; Selinger-Looten et al. 1999). For the downstream reaches with more invader and endurer adaptations prevailing, rehabilitation processes are paramount in biodiversity conservation. The prevailing large-scale processes in downstream reaches, favour endurer species and the development of extended local populations, as was observed. For the Meuse reach furthest downstream,

Figure 6.14 Prospection of the development of protected habitats for a part of the Flemish Common Meuse floodplain with the ECODYN model (Van Braeckel & Van Looy 2005).

an optimization in the sea-closing weir management allows the rehabilitation of tidal impact (Van Leussen et al. 2000; Kerkhofs et al. 2005).
The reaches II, III and IV with higher disorder are not only characterized by higher

river dynamics. Equally important is their location at the confluence of major tributaries and subcatchments as well as ecoregions. This explains the disorder character of reaches II and IV with lower river dynamics but strong influence from large tributaries and adjacent ecoregions.

So, reaches with lower disorder character are more independent of upstream energy, material and propagules. These reaches can be treated isolated from other reaches and uplands/hinterlands, and plans can be elaborated on a more local basis. For instance for the Lorraine Meuse, the contact with adjacent ecoregions which proved very low in species composition, does not prevail in conservation strategies. For reaches with high disorder, the contact with adjacent reaches and regions does matter.

Conclusions for the Common Meuse restoration programme

For the river's flow regime

For general conclusions regarding the flow regime requirements we can refer to literature (Poff et al. 1997) and its formulations: "River managers should address the prevention of unnatural low flows and the preservation - or in the European context, restoration - of the natural flooding regime and morphological dynamics, as a prerequisite for the restoration of semi-natural river corridors which would have major benefits for nature conservation, not only locally but also at the landscape scale" (Petts & Bradley 1997). For the Common Meuse project, the following important aspects were documented and guidelines formulated:

- *mitigation of the hydropeaking pressure;* the potential dampening of peak velocities by bed widening measures is described.
- *restoring contact with the river* for disrupted floodplain zones with important habitats. For these zones the necessary river contact by flooding can be restored either by dike relocation or lowering of artificial levees.
- *restoring gradients in flooding regime* to allow the biocenoses of the dif-

ferent floodplain zones to develop; flood frequency is a driver in community composition and diversity over the river-floodplain system. A variety of communities can develop over the river system if the floodplain gradient is well-distributed over the area.

- *freedom for flood events* to create new habitat in the floodplain; a naturally managed riparian corridor is necessary to allow for sedimentation and erosion processes in the floodplain, essential in the dispersal, recruitment and survival of characteristic habitats, like the dry river grasslands.
- *providing for eroding banks* to feed the river with the necessary sediment for morphological dynamics; although the river has an overload of fine sediments at peak flows, sediment transport of larger particles (sand and gravel) is generally low at this moment, and flushed through the normalized stretches of the Common Meuse. A balanced alternation of widened deposition sections and eroding sections with cut off banks must be developed, to achieve the desired morphological dynamic equilibrium.

For the design and planning of measures

The description of the ecological minimum and natural baseline in the spatial arrangement of habitat templets and ecotopes, can be translated to planning criteria. As principle element for generating this spatial pattern and the ecological integrity of the river ecosystem we documented the morphodynamics and its gradient in the riparian zone and the floodplain. With the modelling approach we provided for a tool in the design of scenarios and optimisation of measures. The disorder is integrated in the modelling in the sense that infrequent floods are a specific step in the model, and heterogeneity is created in the dynamic modelling of different scale processes (topological/chorological) working at a site and it is evaluated in habitat networks.

From our disorder analysis, these general conclusions can be added:

- *heterogeneity and availability of resources for river communities,* essential elements for the riparian biodiversity, develop under natural disturbances (without intervening); river restoration gives the best result when no over-detailed designing and habitat reconstruction (with planting or seeding of desired species). The first and best option in river restoration is more often leaving it up to the river (Kauffman et al. 1997). To the practices of ‘technical’ restoration and enhancements, strong precaution is needed and general guidelines can be for no seeding/planting, unless in a well-argued reintroduction programme.

- no deterministic approach: not in conservation objectives, nor in practices
No overall strategy works at river basin level, nor does it at reach level. Local discontinuities, specific features and local habitat configuration can demand for corrections to the blueprint of restoration measures. The lowering of artificial levees and riverbanks, widening of the river bed and reconnection of side-arms needs a site-specific approach (see Van Looy & De Blust 2002). This does not contrast to the first point, but it stresses the need to integrate local processes/features, as otherwise these can come to discredit the project. No predefined habitat plans, with prescribed substrate and elevation details, are acceptable for river restoration programmes.

- necessary freedom for the river, demands for necessary room for the river;
the spatial extent needs specific guidelines in the present situation of irreversible constraints/impairments to the system; i.e. the meandering character of the Common Meuse can not be restored, nevertheless some freedom is needed to be succesfull; this equilibrium has to be defined in spatial and temporal criteria.
The erodible river corridor (see Piégay et al. 2005) must be defined in its acceptable measures in space and time.

- preserving what is left, but leaving freedom to stochastic events and dynamic processes.
The preservation of relict habitat and populations of threatened species is a prerequisite for the Common Meuse, with many species at the edge of extinction under present threats of fragmentation. Patterns in species presence and seed input/colonization processes should be carefully considered in conservation and restoration strategies in the riparian zone (Middleton 1999). This confirms the importance of spatial connectivity across the river landscape as a key factor for the ecosystem and the community resilience

(Tabacchi et al. 2005). In the light of declining upstream populations of the threatened species of the Common Meuse reach, their preservation in the area is the more stringent.
Infrequent large floods have proven essential in the generation of habitat, the dispersal and recruitment of rare river corridor species and the restoration of fluvial dynamics and heterogeneity in the area. The stochastic character of these events has to be incorporated in plans as beneficial and not as a threat. Of course safety measures must be the first concern, but surely win-win situations can evolve from integrated modelling of hydraulic and ecological developments (see V.2).

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Samenvatting

De centrale vraagstelling voor het onderzoek was: 'hoe bepaalt de rivierdynamiek het leven in en om de rivier?' en 'kunnen we die dynamiek vatten en ermee omgaan in het rivierherstel?'.

Een 'dynamisch evenwicht' van rivieren werd reeds gedefinieerd in termen van morfologie. We trachten dit evenwicht ook in de ecologie van het riviersysteem te definiëren, om zo een bruikbaar instrument te ontwikkelen voor de planvorming en evaluatie van rivierherstelprojecten.
Rivierherstel heeft enkel kans op slagen als een dynamisch evenwicht nagestreefd wordt. Het hydromorfologisch gedefinieerde dynamische evenwicht ligt in de geometrie van de rivier (met maten als verval, bankfull, sinuositeit) en in het afvoerregime (met maten als afvoervariatie, amplitude, pieksnelheid). Het resulteert in specifieke krachten en patronen die optreden in tijd en ruimte. Dit samenspel van krachten en gebeurtenissen bepaalt ook het leven in en om de rivier. Zowel de regionale biodiversiteit, de specifieke levensgemeenschappen van riviertrajecten als de populatiestrategieën van soorten spelen in op deze interacties en zijn aangepast aan de aanwezige dynamiek. Op zoek gaan naar een dynamisch evenwicht betekent dus iets nastreven wat niet vastgesteld kan worden, of toch zeker niet permanent of continu meetbaar is. Het is immers geen echt evenwicht in de zin van een vast te leggen situatie. De grootste fout in rivierherstel is vaak het te statische karakter van de maatregelen. Betonnen stroomgeleiders, grote keien-riffles of gegraven vastgelegde nevengeulen missen vaak hun positieve effecten, zeker op langere termijn bekeken.

Ondanks de specifieke rol voor de stochasticiteit van de sturende processen, de onvoorspelbaarheid en variabiliteit van ontwikkelingen in tijd en ruimte, toch zijn er een heleboel essentiële evenwichten die als leidraad kunnen gelden voor een geslaagd rivierherstel. Het zijn een set van te respecteren maten, een gulden snede, die het specifieke karakter van het riviertraject tot uiting brengen. Deze hebben we trachten te identificeren vanuit een analyse van het riviersysteem vanaf het stroomgebiedsniveau tot op het niveau van de grindbankhabitat van loopkevers. Belangrijke inzichten komen voort uit referentieonderzoek, onderzoek in pilootprojecten en specifiek soortgericht onderzoek. Soortgroepen of kenmerkende soorten van de rivierhabitats zoals oeverloopkevers, Zwarte populier, Bittere wilg, Maasraket of Veldsalie, tonen specifieke aspecten en 'ranges' van het dynamische evenwicht en kunnen cruciale elementen van die dynamiek blootleggen.

Vanuit een conceptuele benadering op landschapsschaal, uitgekristalliseerd in ruimtelijke scenario's voor de Maasvallei, werd ingezoomd op de sturende factoren in het systeem, eerst op niveau van de vallei tot uiteindelijk op het niveau van gemeenschappen en soorten, habitats en microhabitats. Hieruit werden concrete inrichtings- en beheersvoorstellen geformuleerd, om dan weer terug te keren tot op het hogere niveau om een brede evaluatiemethodiek voor het riviersysteem uit te werken.

In hoofdstuk II wordt een conceptueel kader geschetst voor rivierherstel en planvorming op het schaalniveau van het rivierbekken en het landschap. In deze conceptuele benadering van planvorming en opties voor herstel staat de diversiteit en verstoring in het riviersysteem centraal.
Een eerste belangrijke vaststelling betreft de identiteit van de rivier. Deze is essentieel voor het uittekenen van scenario's en het vaststellen van een conceptuele basis voor de beschrijving en het opstellen van herstelstrategieën voor rivierbekkens. Vanuit deze identiteitsbenadering werd geconcludeerd dat grootschalige strategieën op niveau van riviertrajecten geen accurate voorspelling van ontwikkelingen toelaten. Dit is problematisch in de huidige situatie van sterk gefragmenteerde ecosystemen, waarbij alle initiatieven en maatregelen ter bescherming en herstel van biodiversiteit gericht zijn op individuele locaties. Om in overeenstemming te zijn met de huidige wetgeving, is het definiëren van aanwezige natuurwaarden in huidige en toekomstige situatie evenwel vereist op het niveau van de locatie. Het risico bestaat er dan sterk in dat een specifieke locatie, zoals een ecotoop, geïsoleerd van het landschap wordt beschouwd, met alle beheersproblemen van dien. Ecotopen en habitats moeten immers gezien worden als functionele onderdelen van ecologische netwerken. Het voorbeeld van de Grensmaas toonde duidelijk de mogelijkheden van de methodes van netwerkanalyse voor rivierherstelprojecten, zoals ze ook op basis van verdere studies over de volledige Maas tot uitwerking gebracht werden.

Het voorbeeld van de Grensmaas toonde duidelijk dat met eenvoudige data van landgebruik en criteria voor habitatgebruik van kenmerkende soorten, strategieën geformuleerd kunnen worden voor inrichting van het riviergebied rekening houdend met natuurlijke processen. De methodiek van ruimtelijke doelstellingen voor habitats of ecotopen is zeer succesvol in dit verband. In tegenstelling tot een bena-

dering van individuele habitats of ecotopen, wordt het concept van de connectiviteit gebruikt om strategieën uit te tekenen voor natuurherstel. Deze landschapsbenadering is vooral interessant als een geïntegreerde benadering van soorten en processen in de riviercorridor.
Deze benadering werd nog verder conceptueel uitgewerkt in een analyse van de diversiteit van overstromingsgraslanden langsheen de Maas vanuit de processen en samenhang op stroomgebiedsniveau. Riviernetwerken doorsnijden landschappen en voorzien een natuurlijk kader voor beschermings- en herstelprogramma's. We vonden aanwijzingen voor contact tussen de rivier en de doorsneden landschappen in de samenstelling van overstromingsgraslanden. Deze connectiviteit speelt samen met de rivierdynamiek een cruciale rol voor de diversiteit van de riviergemeenschappen. Deze waarnemingen waren de aanleiding om voor het systeemfunctioneren een 'rivierwanorde' concept te definiëren, naar analogie met de natuurkundige wetmatigheden voor de rivier. Met dit concept werden een aantal tekortkomingen in bestaande rivierconcepten aangevuld. Het continuïteitsprincipe van het Rivier Continuüm Concept is immers een te beperkt concept voor keuzes in beschermingsstrategieën voor de rivier, aangezien het alleen continue patronen en processen beschouwd. De wanorde – aanwezig in de combinatie van discontinue processen en extreme heterogeniteit – in de riviercorridor vormt een sterke aanwijzing voor de patronen van biodiversiteit en de wijzigingen in samenstelling langsheen de rivier. Ze wordt veroorzaakt door abrupte wijzigingen in het fysische en biotische milieu van het riviersysteem, die ontstaan door wijzigingen van geomorfologische of geografische oorsprong. De wanorde in de riviercorridor kan dan ook geïdentificeerd worden in functionele of biogeografische kenmerken van soorten/gemeenschappen en kan als leidraad dienen in natuurbeschermingsstrategieën.
Basiselement voor een rivierwanordebenadering is de connectiviteit langsheen én dwars op de rivier met aangrenzende ecoregio's, die van belang is in het behoud van locale biodiversiteit voor riviertrajecten die onder invloed staan van verstoringen en milieugradiënten.

Hoofdstuk III bestaat uit de analyse van sturende factoren in het riviersysteem. Voor het zoeken en vaststellen van de sturende processen achter de biodiversiteit op de verschillende schaalniveaus werden groepen van organismen en gemeenschappen geselecteerd die relevant zijn voor specifieke onderdelen van deze analyse; onderdelen van het rivierlandschap, onderdelen van de samenhang op niveau van het bekken of het traject of een deelgebied.

Een eerste beschrijving van sturende processen in het riviergebied werd gericht op de rivierbossen. Aspecten van samenstelling en diversiteit van de gemeenschappen over de gradiënt van de riviervallei werden gerelateerd aan omgevingsparameters zoals contact met de rivier en isolatie en fragmentatie. De bossen in de Maasvallei die afgesneden zijn van de rivierinvloed verliezen geleidelijk typische riviersoorten en winnen niet evenredig aan typische bosplanten in de tijd. De afgesneden bossen zijn dan ook soortenarmer en minder divers omwille van toenemende dominantie van specifieke soorten. De ecologische verklaring ligt in 1) de beperkte verbreiding van typische bossoorten om de afgesneden bosfragmenten in de vallei te bereiken, en 2) de 'intermediaire verstoring' hypothese die een daling in soortenrijkdom voorspelt bij het ontbreken van verstoring.

Vervolgens werd de invloed van de overstromingskracht op de riviergemeenschappen in beeld gebracht. Hiertoe werden de overstromingsgraslanden onderzocht, met specifieke aandacht voor de droge stroomdalgraslanden en hun bedreigingen in de vorm van isolatie, habitatcreatie en extinctie van zeldzame soorten. De kennis van het proces van habitatcreatie, en van de ruimtelijke en temporele vereisten van de gemeenschappen, laat toe om effectieve beschermingsmaatregelen voor te stellen. De stroomdalsoorten bewijzen een goede gidssoortengroep te vormen voor de beschermings- en herstelinspanningen, aangezien ze informatie bieden over de kwaliteit van aanwezige habitat. Ze komen immers slechts voor in de soortenrijkste en best ontwikkelde standplaatsen. Een modelbenadering gebaseerd op een analyse van de relaties tussen de gemeenschap en de milieuvariabelen (CCA-correspondentieanalyse) bracht beheersrichtlijnen en herstelmogelijkheden aan het licht voor de droge stroomdalgraslanden in het Grensmaasgebied.

Een derde analyse richtte zich op het oevermilieu van de rivier; met een onderzoek naar oeverloopkevers langsheen de volledige Maas. Hier werden sturende processen in het beheer van de rivier gezocht. Onderzoek en beoordelingsinstrumenten voor rivierherstel en hoogwaterbescherming zijn overwegend gericht op hydrologische relaties. Geomorfologische aspecten krijgen pas sinds kort de aandacht die ze verdienen. Vanuit een habitat 'templet' benadering werden de hydromorfologische kenmerken van de rivieroevers in beeld gebracht, gebaseerd op de aanwezige habitat- en soortgroepkenmerken. Naast de algemeen gekende relaties met overstromingsduur en waterdiepte, werden een aantal essentiële parameters afgeleid, zoals breedte-diepte verhouding, habitatdiversiteit en pieksnelheid. Een brede set aan hydrologische en morfologische parameters die de afvoerpiek en de morfodynamiek beschrijven, werd geïdentificeerd om de integriteit van de rivieroever te evalueren.

Hoofdstuk IV behandelt drukken en impact en de gevolgen voor inrichting en beheer zowel voor grotere projecten en trajecten als op niveau van individuele ecotopen en habitats. Voor grote rivieren die overwegend sterk afwijken van de onverstoorde toestand, is het van belang meetbare indicatoren van oorzaak-effectrelaties te kennen om herstel te kunnen voorstellen en evalueren. De respons op drukken werd voor de geselecteerde gemeenschappen geanalyseerd binnen een specifiek schaalniveau.

Voor de bosontwikkeling op de rivieroevers werden de fysische en biologische aspecten onderzocht die een mogelijk herstel sturen. Voor een goede voorspelling van de bosontwikkeling in een modellering is een onderverdeling in tijd-ruimtesequenties een vereiste. Vooral het onderscheiden van een morfodynamische en een biotische component in de ontwikkeling bleek een essentiële stap in de ontwikkelde methode. De analyse bracht een aantal richtlijnen voor rivierherstel naar voor, van toepassing op de verschillende schaalniveaus. Op het niveau van het stroomgebied, moet de voorziening van sediment en van zaden, alsook de connectiviteit en transportcapaciteit geanalyseerd worden. Op het niveau van riviertrajecten, is het voorzien van ruimte en vrijheid voor de rivier essentieel om de bosontwikkeling op gang te brengen en in stand te houden. Een natuurlijk afvoerregime en de nodige morfodynamiek zijn een vereiste op deze schaal. Het herstel van processen is dus de eerste vereiste, eerder dan het richten op ruimtelijk habitatherstel.
Voornamelijk op het locale niveau, dreigt een te concrete doelformulering voor habitats de algemene doelstelling van het herstel van rivierbos, in al zijn tijd-ruimtesequenties, in de weg te staan. Voor een duurzame bosontwikkeling moeten alle tijd-ruimtesequenties afzonderlijk duurzaam aanwezig zijn. Voor het Grensmaasgebied concreet, is de aanwezigheid van zaadbronnen voor Zwarte populier en Bittere wilg problematisch. De verspreiding van de tijd-ruimtesequenties is ook verstoord in de huidige situatie, met slechts enkele oudere bosplekken aanwezig. De Zwarte populier werd geïdentificeerd als

beperkt in vestiging eerder dan in verbreidingsmogelijkheden in de benedenstroomse trajecten van grote rivieren. Voor de Grensmaas bleek bescherming van de aanwezige relicten en een herintroductie vereist om de soort en het typische oeverbostype te herstellen.

De stroomdalplanten van de Maas werden als doelsoortgroep geïdentificeerd en onderzocht naar hun ruimtelijke habitatvereisten in het Grensmaasgebied en specifiek naar de herstelmogelijkheden.
De stroomdalsoorten kwamen naar voor als een goede gidssoortgroep voor de bescherming en herstelinspanningen voor de overstromingsvalleien van de grote Noord-Westeuropese rivieren. Ze geven namelijk een heleboel informatie over de karakteristieken van habitats in deze systemen en tevens vormen ze indicatoren voor de beter ontwikkelde vegetaties van de overstromingsvlakten.
De gewijzigde overstromingsdynamiek van het winterbed is de sterkste bedreiging voor de stroomdalsoorten in de huidige situatie. Relicten die aan de overstromingsdynamiek onttrokken werden, tonen geen herstelmogelijkheden, in tegenstelling tot relicten en soorten op plaatsen met hoge dynamiek. De soorten die door de winterdijken afgesloten zijn van de rivier, hebben een ernstig isolatieprobleem, terwijl het juist als een algemeen gangbare beschermingshypothese voor deze soortengroep geldt. De sterke kolonisatie die we vaststelden voor de bedreigde soorten, doet de balans overslaan naar een keuze voor dynamiekherstel, eerder dan relictbescherming. Desalniettemin is ook het beschermen van die laatste relicten essentieel voor het behoud van die soorten in het gebied. Voor een effectieve herstelstrategie is dus zowel een gewijzigd rivierbeheer als uiterwaardbeheer van belang.

De kritische laagwaterafvoer op de Grensmaas werd onderzocht vanuit de oeverloopkevergemeenschap. Vooral voor de effecten van het piekregime bij de lage afvoeren, ontbraken nog ecologische criteria om het beheer op te richten. Bij de stuw van Borgharen werden reeds maatregelen ondernomen om de sterke fluctuaties, veroorzaakt door de hydroturbines aan de stuw van Lixhe, te dempen. Vanuit de vastgestelde impact op de oeverloopkevergemeenschap werden kritische grenswaarden gedefinieerd voor deze hydromorfologische druk. De resultaten geven aan dat de fluctuaties in het laagwaterregime nog een kwart verder gedempt moeten worden om een aanvaardbare pieksnelheid te bekomen op de Grensmaas. De beoogde rivierherstelprojecten kunnen ook bijdragen aan deze verbetering. De rivierbedverbreding voorzien in het Grensmaasproject, dempt de sterke fluctuaties ook zeer doeltreffend. De eerste locatie benedenstrooms de stuw van Borgharen kan dusdanig uitgetekend worden dat ze de pieken grotendeels opvangt en de rest van het Grensmaasgebied veiligstelt voor de extreme pieksnelheden. Vanuit deze gemeenschap aanwezig in het kritische milieu voor deze druk, werden dus kritische grenswaarden aangegeven, bruikbaar voor het rivierbeheer en het herstelproject.

In hoofdstuk V worden instrumenten voor beoordeling en evaluatie van rivierherstel gepresenteerd op de verschillende schaalniveaus. Om een goede inschatting te kunnen maken van de impact van de mens op de rivier, ontbreekt vaak de kennis van het complexe functioneren van intacte riviersystemen. Vanuit de taakstellingen in de Kaderrichtlijn Water en de zoektocht naar een goede ecologische toestand, gingen we op zoek naar referentiecondities voor de hydromorfologie in relatie tot het biotische systeem. Vanuit een grote dataset van hydrologische en geomorfologische gegevens voor de grote grindrivieren in onze Europese ecoregio, selecteerden we vier vergelijkbare trajecten. Het traject van de Allier bleek goed te voldoen aan de vereisten voor een referentie en hieruit werden referentiecondities voor de grote grindrivieren van de ecoregio afgeleid. De oeverversteviging gaf een belangrijke indicatie naar de impact van menselijke ingrepen op de hydromorfologie in een niet-lineaire respons. De breedte-diepteverhouding was de best bruikbare indicator voor de hydromorfologische toestand omwille van de lineaire respons. Het onderzoek van de oeverbossen resulteerde in een set van kwantitatieve maten voor de referentieomstandigheden en een monitoringvoorstel. Het oeverbos bleek een goede indicator voor de relatie tussen het biotische systeem en de hydromorfologische toestand. De analyse van enkele maten voor oeverbosontwikkeling en de aanwezigheid van doelsoorten Zwarte populier en Bittere wilg, in relatie tot de oeverversteviging en de breedte-diepteverhouding, leverde enkele bruikbare meetlatten voor de beoordeling en evaluatie van herstelprogramma's en beschermingsmaatregelen voor grote rivieren.

Een dynamisch model ECODYN werd ontwikkeld om de mogelijkheden voor rivierherstel te kunnen analyseren en de ecologische doelen te kwantificeren. Met ECODYN volgen we de keuzen en opties die in zwang zijn voor het opmaken van modellen; een meer dynamische aanpak in de expertsystemen waarin tot op heden overwegend statische correlaties toegepast worden, met de kracht van een specifiek model, aangezien modeloplossingen voor specifieke problemen, bruikbare elementen kunnen aanleveren voor complexere modellen. Bovendien berust het model op een aantal pragmatische benaderingen gebaseerd op eenvoudige empirische relaties aangevuld met expertkennis, eerder dan complexe simulatiemodellen van ecosysteemprocessen. In het model werd de expertise met betrekking tot bosontwikkeling, graslandontwikkeling en oeverhabitats in specifieke modules ingepast en samengesmeed tot een geïntegreerde voorspelling van sturende processen voor ecotoopontwikkeling over het riviergebied. De resultaten voor het Grensmaasproject pakten positief uit voor de ruwheidsdoorrekening en hoogwaterberekeningen en gaven nieuw perspectief aan het vraagstuk van stroomweerstand en natuurontwikkeling. Het gaf tevens een bijkomende stimulans om met meer accurate modelleringen te gaan werken voor het verdere planontwerp.

In een internationale samenwerking werd een uitgebreide evaluatiemethodiek voor hoogwaterbeschermingsstrategieën uitgewerkt op stroomgebiedsniveau, opgehangen aan het concept van het ‘ecologische minimum’. Drie scenario’s op stroomgebiedsniveau werden uitgetekend, gebaseerd op onderscheiden maatregelen voor sponswerking, retentiecapaciteit en winterbedmaatregelen. Een methodiek werd ontwikkeld voor de ecologische effectinschatting op verschillende schaalniveaus en voor verschillende componenten van het rivierecosysteem, namelijk de ruimtelijke configuratie van habitats, de overstromingsgraslanden in het winterbed en de loopkevers in de rivieroever. Voor de overstromingsgraslanden werd een beoordeling op basis van overstromingsfrequentie en de hydrologische gradiënt in het winterbed opgemaakt. Voor de loopkevers en de oeverzone werd de respons en beoordeling vastgesteld op 1) de kenmerken van afvoerpieken (pieksnelheid en piekfrequentie), en 2) de variatie in de oever, meetbaar in de breedte-diepteverhouding. Deze factoren zijn ook duidelijke invloedsfactoren van hoogwaterbeschermingsstrategieën en om deze reden bieden ze een bruikbaar kader voor een evaluatiemethode. Het definiëren van een ecologisch minimum werd voorgesteld vanuit het praktische oogpunt van het huidige beleidskader rond rivieren, dat streeft naar een ecologisch herstel, maar hiertoe goede maatlatten mist. Voor grotere rivieren en overstromingsgebieden is de integratie van hoogwaterbescherming en rivierherstel een grote uitdaging, en het definiëren van een venster tussen een ecologisch minimum en optimum, een zeer bruikbaar instrument.

De synthese in hoofdstuk VI wordt opgebouwd rond het rivierwanorde concept en de toepassing ervan in het rivierherstel. Het accuraat formuleren van doelstellingen is al te vaak het grote probleem voor rivierherstelprojecten. Het gebrek aan kennis van de menselijke verstoring en de druk-impact relaties voor het biotisch systeem is hiervoor de belangrijkste leemte. Een bijkomend probleem is de afstand tussen het wetgevende kader en de specifieke eisen van elke rivier of traject van rivier. Vanuit het rivierwanorde concept schetsen we een kader voor het opbouwen van de nodige kennis om goede beschermingsdoelstellingen te formuleren.
Basis voor het rivierwanorde concept vormt het dynamische evenwicht aanwezig in het riviersysteem, eigenlijk een toestand van onevenwicht veroorzaakt doordat de kenmerken van de rivier steeds een samenspel en momentopname zijn in het geheel van geomorfologische, geografische en klimatologische ontwikkelingen. Dit onevenwicht vinden we tevens terug in het biotische systeem tot op het niveau van gemeenschappen die los van structuur zijn en niet gekenmerkt door saturatie. Dit dynamische evenwicht heeft voor gevolg dat in de aanpak voor bescherming en herstel geen statische, deterministische doelen geformuleerd kunnen worden en dat er tevens voor elke specifieke locatie, riviertraject en rivier afzonderlijke doelstellingen vereist zijn, dynamisch zowel in ruimtelijke als in temporele zin.
Specifieke doelstellingen en richtlijnen voor de Maas en meer specifiek voor de Grensmaas werden afgeleid met de rivierwanordebenadering, vanuit discontinuïteiten, sterke gradiënten, heterogeniteit en onvoorspelbaarheid. Ze worden beschreven in een doelstellingbenadering met een 'guiding image', niet zozeer een 'Leitbild'-aanpak als wel een aanpak vanuit een samenspel van richtlijnen en vereisten vanuit zowel soorten en levensgemeenschappen als ruimtelijke kenmerken en fysische processen.

Niet zozeer het scheppen van een herhaling van een specifiek referentiebeeld, of de bescherming van specifieke aandachtsoorten, of de aanwezigheid van adaptaties in aanwezige gemeenschappen, werden onderscheiden als het belangrijkst in het opmaken van rivierherstelprojecten, maar wel de patronen die ontstaan zijn door stochastische processen, door interacties van biotische en abiotische processen of door combinaties van gradiënten die longitudinaal en lateraal werken in het riviersysteem.

Appendix table S1:

Plant species matrix for the Meuse reaches floodplain meadow vegetation sampling, with frequency score over the plots for every reach (in Tansley score, Tansley, 1935), classifications and correspondence analysis score.

Species	I	II	III	IV	V	VI	OECO _GRP*	Eco† region	AMPL	Dist_ adapt	Pop‡dyn	DCA1	DCA2	DCA3
Achillea millefolium	f	f	f	f	f	f	51	3	1	3	E	166.83	186.56	156.44
Achillea ptarmica	r	r	f	r	r	r	52	1	2	3	P	143.00	189.42	151.13
Aconitum vulparia		r					91	4	1	4	SS	116.22	297.43	284.50
Acorus calamus		r	s	r	o		43	1	1	2	SC	145.92	173.03	165.33
Aethusa cynapium		s	f	r	r		11	3	1	3	SC	138.36	185.42	158.74
Agrimonia eupatoria		r	f	r			83	4	2	4	P	114.17	225.06	137.66
Agrostis gigantea			f	f	o	r	21	1	1	3	E	132.61	133.30	116.81
Agrostis stolonifera	f	f	f	f	f	f	21	1	1	2	E	166.83	186.56	156.44
Agrostis capillaris			f	f	f	f	64	2	1	4	E	148.34	113.70	133.56
Aira caryophyllea		r		s			64	2	2	4	R	133.88	252.26	189.67
Aira praecox					s		64	2	1	4	R	198.84	86.32	211.45
Ajuga reptans	f	f	f	f	f	f	52	1	1	2	E	166.83	186.56	156.44
Alisma lanceolatum		r	o	r			43	1	2	1	SC	81.55	219.77	132.41
Alisma plantago-aquatica	f	f	f	f	f	f	43	1	2	1	P	166.83	186.56	156.44
Allium oleraceum		r	r	s			83	4	2	3	R	80.33	233.98	161.52
Allium scorodoprasum		s	s				93	1	2	3	R	58.11	251.99	201.90
Allium vineale		o	f	r			82	3	2	4	P	76.34	226.93	147.86
Alopecurus aequalis					s	s	22	1	1	1	R	212.08	43.16	207.47
Alopecurus geniculatus	f		o	f	f	f	21	1	2	2	P	186.26	162.16	131.43
Alopecurus myosuroides		r	r	s			11	3	1	1	M	80.33	233.98	161.52
Alopecurus pratensis	f	f	f	f	f	f	51	3	1	3	E	166.83	186.56	156.44
Althaea officinalis					s	o	44	1	2	3	P	218.70	21.58	205.49
Angelica archangelica		r	s	r	o	o	44	1	2	2	SC	167.58	125.84	175.74
Angelica sylvestris	f	f	f	f	f	f	44	1	2	3	P	166.83	186.56	156.44
Anthemis arvensis	o						13	2	1	1	SS	291.38	367.11	119.86
Anthoxanthum odoratum	f	o	f	r	r		51	3-5	2	4	E	150.02	245.56	148.87
Anthoxanthum aristatum		s					13	4	1	4	SS	116.22	297.43	284.50
Anthriscus sylvestris	f	f	f	f	f	f	82	3	2	3	E	166.83	186.56	156.44
Anthyllis vulneraria	o	o	r	s			63	4	1	4	R	154.67	285.41	161.30
Aphanes arvensis		r	s				12	4	2	1	SC	77.48	267.14	229.43
Aphanes inexpectata			o	r	r		13	2	2	4	SS	105.15	159.46	111.54
Arabidopsis thaliana			r	r		o	62	3	1	4	P	144.91	105.29	121.30
Arabis hirsuta		s					63	4	2	4	R	116.22	297.43	284.50

Arctium lappa		o	f	f	f	f	17	1	2	2	E	143.27	142.71	157.39
Arctium minus		o	f	f	o	o	17	1	1	1	E	135.17	154.42	151.50
Arctium tomentosum		r	s				17	1	2	1	SC	77.48	267.14	229.43
Arenaria serpyllifolia			r	o	r		62	3	2	4	P	129.32	153.08	94.50
Aristolochia clematitis		r	r				83	4	1	3	R	58.11	251.99	201.90
Arrhenatherum elatius	f	f	f	f	f	f	51	3	1	3	E	166.83	186.56	156.44
Artemisia absinthium		s	s				15	1	1	1	SS	58.11	251.99	201.90
Artemisia campestris					s		62	3	1	1	SS	198.84	86.32	211.45
Artemisia vulgaris	f	f	f	f	f	f	17	1	1	1	E	166.83	186.56	156.44
Aster lanceolatus	r	r	r	o	r	r	44	1	2	3	SC	167.01	184.66	144.40
Aster tripolium					r	o	31	1	1	3	R	214.73	34.53	206.68
Astragalus glycyphyllos	r	r	s				83	4	1	4	M	163.04	307.13	185.61
Atriplex prostrata	r	r	f	r	r	f	15	1	2	1	SC	153.29	165.74	157.68
Ballota nigra foetida		r	r				17	4	2	3	SC	58.11	251.99	201.90
Barbarea vulgaris	r	r	o	r			44	1	2	1	P	128.18	252.51	129.62
Bellis perennis	f	f	f	f	f	f	51	3-5	2	4	E	166.83	186.56	156.44
Bidens cernua		r	f	f	f	f	22	1	1	1	E	144.77	134.12	150.33
Brassica nigra	f	f	f	o	o	o	44	1	2	1	P	162.40	201.39	159.02
Briza media	f	r	s	s			51	3	1	4	P	195.90	303.97	145.97
Bromus erectus	o						63	4	1	4	P	291.38	367.11	119.86
Bromus hordeaceus	f	f	f	o	r	r	51	3	1	4	E	157.18	218.04	153.92
Bromus racemosus	o						52	1	1	4	P	291.38	367.11	119.86
Bromus sterilis			f	r			82	3	2	3	SC	56.40	191.69	79.53
Bromus tectorum		r	r				16	4	2	1	SC	58.11	251.99	201.90
Bryonia cretica	o	o	o				84	5	1	3	P	135.87	290.36	174.55
Bunias orientalis		s	s				17	1	1	1	SS	58.11	251.99	201.90
Calamagrostis canescens			r	r	r		71	2	2	3	P	122.68	151.60	110.25
Calamagrostis epigejos						r	81	3	2	3	E	225.33	0.00	203.50
Caltha palustris	o				r	r	52	1	2	3	R	246.07	181.99	169.93
Calystegia sepium	f	f	f	f	f	f	44	1	2	3	E	166.83	186.56	156.44
Campanula glomerata						s	63	4	2	4	M	225.33	0.00	203.50
Campanula rapunculus		r	f	r	s		83	4	1	4	P	85.52	203.48	139.74
Campanula rotundifolia	f	r	f	r	s		64	2	2	3	E	148.86	253.82	133.62
Campanula trachelium		r	s				94	4	1	3	M	77.48	267.14	229.43
Cardamine hirsuta	r	o	f	f	o	r	62	3-5	2	1	E	147.52	186.64	145.10
Cardamine impatiens		r	o	r	r		91	3	1	3	P	107.61	190.12	149.98
Cardamine pratensis	f	f	f	f	f	f	51	3-5	2	3	E	166.83	186.56	156.44
Carduus crispus	r	r	f	o		o	17	1	1	1	E	142.77	188.65	135.46
Carduus nutans	r		s			r	16	4	2	1	M	206.68	188.16	153.21
Carex acuta	f	f	f	f	f	f	43	2	2	2	E	166.83	186.56	156.44

Carex acutiformis			r	r	r	r	43	2	1	3	P	148.34	113.70	133.56
Carex arenaria				s	r		62	3	1	2	P	188.96	111.52	140.96
Carex caryophyllea		r	s	s	s		63	4	1	4	M	120.10	209.93	179.95
Carex disticha	f	s	s			r	52	1	1	3	E	216.55	246.55	161.28
Carex flacca		s	s				72	4	1	3	M	58.11	251.99	201.90
Carex hirta	f	f	f	f	o	o	21	1	2	2	E	162.71	199.59	151.80
Carex elata		s	s	r	s		43	2	2	3	P	130.69	182.84	123.05
Carex cuprina	o	r	f	o	o	o	21	1	2	3	E	160.37	181.51	147.26
Carex ligerica						r		1	1	3	P	225.33	0.00	203.50
Carex ovalis		r	r				21	1	1	3	P	58.11	251.99	201.90
Carex panicea	o						73	5	1	4	P	291.38	367.11	119.86
Carex riparia	r	o	o	o	o	o	43	1	2	2	E	159.50	175.94	158.59
Carex spicata		r	f	f			82	3	2	3	E	90.92	206.88	104.62
Carex tomentosa	o						73	5	1	3	R	291.38	367.11	119.86
Carex vesicaria	o		r	r	r	r	43	1	2	3	P	187.35	182.81	129.83
Carex vulpina	o						21	1	1	3	R	291.38	367.11	119.86
Carlina vulgaris		s			s	s	63	4	2	4	M	180.13	127.91	233.15
Centaurea jacea	f	f	f	f	f	o	21	1	2	3	E	164.28	194.67	154.39
Centaurea scabiosa		s	s				63	4	1	4	M	58.11	251.99	201.90
Centaurium erythraea			o		r		81	3	2	4	P	79.54	158.46	156.16
Centaurium pulchellum			r				23	2	1	3	R	0.00	206.56	119.30
Cerastium arvense			o	o	r		62	3	2	4	P	113.16	159.77	97.60
Cerastium brachypetalum			s				62	3	1	4	M	0.00	206.56	119.30
Cerastium glomeratum	r	r	o	r	r		15	1	2	1	P	141.03	222.30	144.50
Cerastium fontanum vulgare	f	f	f	f	f	f	51	3-5	1	3	E	166.83	186.96	156.44
Cerastium pumilum			s				62	3	2	1	M	0.00	206.56	119.30
Cerastium semidecandrum			r	o	o	r	62	2	2	1	P	155.48	115.79	127.99
Chaenorrhinum minus	r	r	r				12	5	2	1	SC	135.87	290.36	174.55
Chaerophyllum temulum			f				82	3	2	4	E	0.00	206.56	119.30
Chenopodium bonus-henricus	f	s	s				17	3-5	2	1	SS	213.62	328.74	147.21
Chenopodium polyspermum	o	o	f	o	o	o	11	1	2	1	E	158.05	187.61	154.48
Chenopodium rubrum			f	r	r	r	22	1	2	1	E	118.67	132.27	130.71
Leucanthemum vulgare	f	o	f	f	o	o	51	3	2	3	E	164.93	194.94	145.48
Cichorium intybus			f				51	3	2	3	P	0.00	206.56	119.30
Circaea lutetiana		r	r				91	3	2	4	P	58.11	251.99	201.90
Cirsium arvense	f	f	f	f	f	f	17	1	2	1	E	166.83	186.56	156.44
Cirsium oleraceum	o	r	s				52	1	2	3	P	184.43	317.12	174.65
Cirsium vulgare	f	f	f	f	f	f	15	3	1	1	E	166.83	186.56	156.44
Clematis vitalba		r	o	r			84	5	2	3	E	81.55	219.77	132.41
Colchicum autumnale	f	s	s				52	1	1	3	R	213.62	328.74	147.21

Conium maculatum			r	r	r	r	17	1	2	3	M	148.34	113.70	133.56
Convolvulus arvensis	f	o	f	f	r	r	15	3-5	2	1	E	159.96	210.91	138.95
Cornus sanguinea	o	o	o	r			84	5	2	4	R	141.93	267.01	142.82
Coronilla varia	s	s	s				63	4	2	3	M	135.87	290.36	174.55
Coronopus squamatus	o	o	o	o	o	o	14	1	2	1	SC	166.83	186.56	156.44
Corydalis solida	r	r	r				93	1	1	3	P	135.87	290.36	174.55
Corynephorus canescens			s	s			64	2	1	4	M	84.60	184.25	59.65
Crataegus monogyna	f	f	f	r	r	r	84	5	2	3	E	156.51	221.16	162.48
Crepis biennis	f	r	f	r		r	51	1	2	3	E	156.21	229.53	138.05
Crepis capillaris			o	r			15	3	2	4	E	67.68	188.71	71.58
Crepis vesicaria			r				51	3	2	3	P	0.00	206.56	119.30
Cuscuta europaea		r	f	r			44	1	2	2	E	71.35	218.12	130.77
Cynodon dactylon		r	o	o	r	f	62	3	2	1	SC	145.64	133.78	154.56
Cynosurus cristatus	f	r	o	r	r		51	3-5	1	4	E	164.16	244.57	140.71
Dactylis glomerata	f	f	f	f	f	f	51	3	1	2	E	166.83	186.56	156.44
Daucus carota	f	f	f	o	f	o	51	3	2	1	E	164.06	196.16	161.41
Deschampsia cespitosa	f	r	r	r	o	o	21	1	2	3	E	188.05	191.20	158.24
Dianthus armeria		r	r				83	4	2	3	M	58.11	251.99	201.90
Dipsacus pilosus		s	o				81	3	2	1	SC	29.06	229.28	160.60
Dipsacus fullonum		r	f	r			16	4	2	1	P	71.35	218.12	130.77
Thelypteris palustris			s	s	s		71	2	2	3	R	122.68	151.60	110.25
Echinops exaltatus		s					16	1	1	3	M	116.22	297.43	284.50
Echium vulgare		r	r				16	4	2	1	P	58.11	251.99	201.90
Eleocharis acicularis			s	s			42	2	2	1	R	84.60	184.25	59.65
Eleocharis palustris	f	r	f	f	f	f	43	1	1	1	R	171.43	176.48	144.79
Elymus repens	f	f	f	f	f	f	15	1	2	2	E	166.83	186.56	156.44
Epilobium ciliatum			r	r	r	r	17	1	2	1	E	148.34	113.70	133.56
Epilobium hirsutum	f	f	o	f	f	f	44	1	1	3	P	174.08	185.69	158.05
Epilobium lanceolatum			f	r	r		81	3	1	1	E	92.01	165.34	112.51
Epipactis helleborine			r	r			92	2	1	3	P	84.60	184.25	59.65
Equisetum arvense	f	f	f	f	f	f	15	3-5	2	3	E	166.83	186.56	156.44
Equisetum fluviatile	f		s	s			43	2-5	2	3	R	222.45	306.16	99.79
Equisetum palustre	f	s				r	21	1	1	3	P	247.49	252.27	167.28
Erigeron acris			o	r			62	3	2	1	SS	67.68	188.71	71.58
Erodium cicutarium			r	r	r	r	13	2	2	4	P	148.34	113.70	133.56
Erophila verna			r	r	r	r	62	3	2	4	E	148.34	113.70	133.56
Eryngium campestre			o	r	r	o	63	1	2	2	P	141.20	111.62	139.13
Erysimum cheiranthoides	f	r	f	r			11	1	2	1	P	144.70	267.78	127.14
Eupatorium cannabinum	o	o	o	o	o	o	44	1	2	3	E	166.83	186.56	156.44
Euphorbia cyparissias		r	s				63	4	2	3	M	77.48	267.14	229.43

Euphorbia esula esula		o	f	o	r	r	16	1	2	3	E	121.76	169.78	154.33
Euphorbia seguieriana				s		s	63	4	2	3	M	197.26	80.97	101.75
Odontites vernus			f				21	1	2	3	E	0.00	206.56	119.30
Festuca arundinacea	f	f	f	o	o	f	21	1	2	3	P	165.26	192.23	161.05
Festuca pratensis	f	f	f	f	f	f	51	1	2	3	E	166.83	186.56	156.44
Festuca rubra	f	f	f	f	f	f	51	1	2	4	E	166.83	186.56	156.44
Filago minima			s				64	4	2	4	M	0.00	206.56	119.30
Filipendula ulmaria	f	f	f	f	f	f	52	1	2	3	E	166.83	186.56	156.44
Fumaria officinalis		r	o				11	3	2	1	SC	46.49	242.91	185.38
Galeopsis ladanum angustifolia		s	s				12	4	1	4	R	58.11	251.99	201.90
Galeopsis tetrahit	f	r	f	r		r	82	3-5	2	3	E	156.21	229.53	138.05
Galium aparine	f	f	f	f	f	f	82	3-5	1	2	E	166.83	186.56	156.44
Cruciata laevipes		r	f	r			82	3-5	2	3	P	71.35	218.12	130.77
Galium mollugo	f	o	f		r		51	1	2	3	E	147.07	258.43	171.77
Galium palustre elongatum	o						43	5	1	3	P	291.38	367.11	119.86
Galium verum	f	r	f	o	o	o	62	3-5	2	3	P	167.27	191.28	145.82
Geranium columbinum			s				82	3	2	4	SC	0.00	206.56	119.30
Geranium dissectum	f	f	f	f	f	f	11	1	2	1	E	166.83	186.56	156.44
Geranium molle			f	f	f	f	15	3-5	2	3	P	148.34	113.70	133.56
Geranium pusillum			s	s		r	15	3-5	2	4	SS	154.96	92.12	131.58
Geranium pyrenaicum		s	o				17	4	1	4	P	29.06	229.28	160.60
Geranium rotundifolium		s	r				15	4	2	1	M	38.74	236.85	174.37
Glyceria maxima	f	f	f	f	f	f	43	1	2	2	E	166.83	186.56	156.44
Gnaphalium uliginosum	f	f	f	f	f	f	23	1	2	1	E	166.83	186.56	156.44
Gratiola officinalis	r						21	1	1	3	R	291.38	367.11	119.86
Avenula pubescens	o	s	r	r	r	r	63	4	1	4	P	181.42	192.36	142.72
Heracleum sphondylium	f	f	f	f	f	f	82	3-5	2	3	E	166.83	186.56	156.44
Herniaria glabra		r	r	r	r	s	21	1	1	4	R	132.65	167.16	159.33
Hieracium vulgatum		s	s				95	5	2	4	M	58.11	251.99	201.90
Hieracium laevigatum		s	s				95	5	2	4	M	58.11	251.99	201.90
Hieracium pilosella			r	f	f	o	62	3	2	4	P	165.24	108.16	130.38
Hieracium umbellatum		o	f				95	5	2	4	P	49.81	245.50	190.10
Holcus lanatus	f	f	f	f	f	f	51	1	2	3	E	166.83	186.56	156.44
Hordeum murinum		r	f	f	f	f	14	1	2	1	E	144.77	134.12	150.33
Hordeum secalinum	f		r				51	3-5	2	4	SC	194.25	313.59	119.67
Humulus lupulus	f	f	f	f	f	f	84	5	2	3	E	166.83	186.56	156.44
Hyoscyamus niger		s	r				16	4	1	1	SC	38.74	236.85	174.37
Hypericum perforatum			f	f	r		64	2	2	3	E	107.45	164.66	90.01
Hypericum quadrangulum		r	r	r	r	r	52	1	1	3	R	141.92	150.45	163.75

Hypochaeris radicata			f	f	f	f	62	3	2	4	P	148.34	113.70	133.56
Inula britannica	o		r		r	r	21	1	2	2	R	191.39	187.45	158.68
Inula conyzae		o	r				83	4	2	3	M	69.73	261.08	218.42
Iris pseudacorus	f	f	f	f	f	f	43	1	2	1	E	166.83	186.56	156.44
Juncus articulatus	f	f	f	f	f	f	21	1	2	1	E	166.83	186.56	156.44
Juncus bufonius	f	f	f	f	f	f	22	1	2	1	E	166.83	186.56	156.44
Juncus compressus	f	f	f	f	f	f	21	1	1	1	P	166.83	186.56	156.44
Juncus conglomeratus			s	r			73	2	1	3	R	112.80	176.81	39.77
Juncus effusus	f	f	f	f	f	f	21	1	2	1	E	166.83	186.56	156.44
Juncus gerardi						o	33	1	1	3	P	225.33	0.00	203.50
Juncus inflexus	f	r	f	f	f	f	21	1	2	3	E	171.43	176.48	144.79
Knautia arvensis	f	o	f	r			51	4	2	4	P	142.51	270.06	139.24
Koeleria pyramidata	o						63	4	2	4	M	291.38	367.11	119.86
Petrorhagia prolifera		s	s				63	4	2	4	SC	58.11	251.99	201.90
Lactuca serriola	r	r	f	r	r		16	1	2	1	E	129.27	220.98	142.40
Lamium maculatum		o	f	o	r		82	1	2	2	P	104.49	198.08	146.13
Lathyrus nissolia			o				83	4	1	3	M	0.00	206.56	119.30
Lathyrus pratensis	f	f	f	f	f	f	51	1	2	3	E	166.83	186.56	156.44
Leersia oryzoides		s	r	s			22	1	2	1	SC	71.35	218.12	130.77
Leontodon autumnalis	f	f	f	f	f	f	21	1	2	3	E	166.83	186.56	156.44
Leontodon hispidus	f	r	o		r		63	4	1	4	P	163.24	259.60	166.29
Lepidium campestre			r				15	3	2	1	SC	0.00	206.56	119.30
Cardaria draba			o	r			15	1	2	1	SC	67.68	188.71	71.58
Lepidium latifolium			r				44	1	2	2	SS	0.00	206.56	119.30
Lepidium ruderale		r	r	r	r		14	1	2	1	SS	121.07	188.06	153.81
Linaria arvensis			s				13	4	1	1	M	0.00	206.56	119.30
Cymbalaria muralis		o	o				61	5	2	1	M	58.11	251.99	201.90
Kickxia elatine		r	r				12	4	2	1	M	58.11	251.99	201.90
Kickxia spuria		r	r				12	4	2	1	M	58.11	251.99	201.90
Linaria vulgaris	f	f	f	r		r	15	3	2	3	P	151.22	238.02	156.35
Lolium perenne	f	f	f	f	f	f	14	1	2	1	E	166.83	186.56	156.44
Lonicera periclymenum		r	r				95	5	1	4	R	58.11	251.99	201.90
Lonicera xylosteum		s					94	4	1	4	M	116.22	297.43	284.50
Lotus corniculatus	f	f	f	f	f	f	62	3-5	2	3	E	166.83	186.56	156.44
Lunaria rediviva		r					82	1	1	1	SS	116.22	297.43	284.50
Luzula campestris			r	f	o	r	64	2	1	4	P	156.72	119.98	116.36
Lychnis flos-cuculi	f	r	r	r	r		52	1	1	3	P	177.84	247.74	142.50
Anchusa arvensis			o	o			13	2	2	1	SC	84.60	184.25	59.65
Lycopus europaeus	f	f	f	f	f	f	43	1	2	1	E	166.83	186.56	156.44
Lysimachia nummularia	f	r	o	r	r	r	21	1	2	3	P	172.31	211.96	149.08

Lysimachia vulgaris	f	f	f	f	f	f	52	1	2	3	E	166.83	186.56	156.44
Lythrum salicaria	f	f	f	f	f	f	44	1	2	1	E	166.83	186.56	156.44
Malva alcea		r	r				17	4	2	3	P	58.11	251.99	201.90
Malva moschata		r	r				82	3	2	4	R	58.11	251.99	201.90
Malva pusilla			s				17	4	2	1	SS	0.00	206.56	119.30
Matricaria maritima	f	f	f	f	f	f	15	1	2	1	E	166.83	186.56	156.44
Medicago falcata		r	r	r		r	63	4	2	4	R	127.69	166.48	151.83
Medicago lupulina	f	o	f	r		r	51	3-5	2	1	P	153.55	234.05	147.81
Medicago minima			s			s	62	3	2	1	SS	112.66	103.28	161.40
Silene latifolia		r	o				15	3	2	3	R	46.49	242.91	185.38
Silene noctiflora		r	r				12	4	2	4	R	58.11	251.99	201.90
Silene dioica		r	o	o	o	o	82	3	2	3	E	143.75	139.95	155.12
Melilotus altissima		r	o	r			44	1	2	3	P	81.55	219.77	132.41
Mentha aquatica	f	o	f	f	f	f	43	2	2	1	E	169.03	181.74	150.87
Mentha arvensis	f	f	f	f	f	f	21	1	2	3	E	166.83	186.56	156.44
Mentha pulegium	o	r	s				21	1	2	1	P	184.43	317.12	174.65
Mentha suaveolens			f	r			21	1	2	3	P	56.40	191.69	79.53
Mentha x verticillata			f	f	r		43	2	1	3	P	107.45	164.66	90.01
Myosotis laxa	f	o	f	f	f	f	21	1	1	1	E	169.03	181.74	150.87
Myosotis ramosissima		r	o		s		62	3	2	4	R	71.88	216.81	189.72
Myosotis palustris			f	f	f	f	43	2	2	1	E	148.34	113.70	133.56
Stellaria aquatica	f	f	f	f	f	f	22	1	2	1	E	166.83	186.56	156.44
Oenanthe aquatica	f						43	2	2	1	P	291.38	367.11	119.86
Oenanthe fistulosa	f						43	2	1	3	P	291.38	367.11	119.86
Oenanthe silaifolia	o						51	1	1	3	P	291.38	367.11	119.86
Oenothera biennis		r	o				16	4	2	1	SC	46.49	242.91	185.38
Oenothera erythrosepala		r	o				16	4	1	1	SC	46.49	242.91	185.38
Onobrychis viciifolia	o						63	4	1	4	M	291.38	367.11	119.86
Ononis repens spinosa		r	f	o	o	r	51	4	2	3	P	127.66	154.70	149.11
Origanum vulgare		r	f	r	o		83	4	1	3	P	106.12	182.17	152.78
Ornithogalum umbellatum	f	r	f	r			82	1	2	2	E	144.70	267.78	127.14
Ornithopus perpusillus					s		64	2	2	4	M	198.84	86.32	211.45
Orobanche minor			o				51	3	1	4	R	0.00	206.56	119.30
Papaver argemone		o	r				13	4	2	1	SS	69.73	261.08	218.42
Papaver dubium						r	13	2	2	1	SS	225.33	0.00	203.50
Papaver rhoeas		f	f	r			11	3	2	1	SS	80.33	233.98	161.52
Parietaria judaica		o	r				61	5	1	1	M	69.73	261.08	218.42
Pastinaca sativa urens		r	o				51	3	2	3	R	46.49	242.91	185.38
Petasites hybridus		r	r	r	o	o	44	1	1	1	E	153.61	132.57	171.04
Peucedanum carvifolia	f	r	s			s	51	1	2	3	P	202.91	283.73	171.41

Phalaris arundinacea	f	f	f	f	f	f	43	1	2	3	E	166.83	186.56	156.44
Phleum pratense pratense	f	f	f	f	f	f	51	1	2	1	E	166.83	186.56	156.44
Phragmites australis	f	o	r	o	o	f	43	1	2	3	E	185.24	185.19	158.94
Picris echioides		s	s				16	4	2	4	M	58.11	251.99	201.90
Picris hieracioides		r	f				63	4	2	3	P	38.74	236.85	174.37
Pimpinella saxifraga	f	o	f		r		62	3	2	4	P	147.07	258.43	171.77
Plantago major pleiosperma		r	o	o	o	o	23	1	2	1	SC	143.75	139.95	155.12
Plantago lanceolata	f	f	f	f	f	f	51	3-5	2	3	E	166.83	186.56	156.44
Plantago major	f	f	f	f	f	f	14	1	2	1	E	166.83	186.56	156.44
Plantago media	f	o	o	r		r	63	4	2	3	P	164.52	236.02	149.85
Poa annua	f	f	f	f	f	f	14	1	2	1	E	166.83	186.56	156.44
Poa bulbosa		s	s				62	3	2	2	R	58.11	251.99	201.90
Poa compressa		f	o				63	4	2	3	P	66.41	258.48	213.70
Poa pratensis	f	f	f	f	f	f	51	1	2	4	E	166.83	186.56	156.44
Poa trivialis	f	f	f	f	f	f	21	1	2	3	E	166.83	186.56	156.44
Polygala vulgaris		s			s		64	5	1	3	M	157.53	191.87	247.97
Polygonum amphibium	f	f	f	f	f	f	21	1	2	1	E	166.83	186.56	156.44
Polygonum aviculare	f	f	f	f	f	f	14	1	2	1	E	166.83	186.56	156.44
Polygonum lapathifolium		r	o	r	r	o	15	1	2	1	E	137.04	142.59	163.36
Polygonum mite		r	o	r	r	r	22	1	2	1	SC	129.02	155.55	159.71
Polygonum persicaria	f	f	f	f	f	f	11	1	2	1	E	166.83	186.56	156.44
Potentilla anserina	f	f	f	f	f	f	21	1	2	2	E	166.83	186.56	156.44
Potentilla argentea			s		s		64	2	2	4	R	99.42	146.44	165.37
Potentilla erecta				s			75	2	2	4	M	169.19	161.94	0.00
Potentilla intermedia			r				15	3	2	4	SS	0.00	206.56	119.30
Potentilla reptans	f	o	f	o	o	f	21	1	2	2	E	291.38	367.11	119.86
Potentilla verna		r	s	s	r		63	4	1	4	M	133.22	189.33	185.20
Primula veris	f	o	o	s			63	4	2	3	R	153.04	285.67	153.71
Prunella vulgaris	f	o	f			r	51	1	2	3	P	151.14	245.15	170.55
Prunus spinosa	o	o	o	s	s		84	5	1	3	R	144.62	260.14	162.04
Pulicaria dysenterica	f	f	f	f	f	f	21	1	2	1	E	166.83	186.56	156.44
Ranunculus acris	f	f	f	f	f	f	51	1	2	1	E	166.83	186.56	156.44
Ranunculus bulbosus	f	f	f	f	f	f	62	4	2	3	E	166.83	186.56	156.44
Ranunculus ficaria		o	f				92	5	2	2	E	49.81	245.50	190.10
Ranunculus repens	f	f	f	f	f	f	21	1	2	1	E	166.83	186.56	156.44
Raphanus raphanistrum			o	r	r		13	3	2	1	SC	105.15	159.46	111.54
Reseda lutea		f	f		r		16	4	2	1	P	86.26	218.86	203.81
Reseda luteola		f	f	r	r		16	4	2	1	P	100.08	209.37	169.84
Rhinanthus alectorolophus			s				51	3	2	4	M	0.00	206.56	119.30
Rhinanthus minor			r	r			75	2	2	4	R	84.60	184.25	59.65

Rorippa amphibia	f	o	o	o	o	f	43	1	2	1	E	175.98	186.26	156.96
Rorippa palustris	o	r	o	o	o	o	22	1	2	1	P	169.80	180.04	148.90
Rorippa sylvestris	f	f	f	f	f	f	21	1	2	1	E	166.83	186.56	156.44
Rubus caesius		o	f	r			84	5	2	3	E	76.34	226.93	147.86
Rumex acetosa	f	f	f	f	f	f	51	1	2	3	E	166.83	186.56	156.44
Rumex acetosella			s	f	r		64	2	1	4	P	153.49	146.71	77.46
Rumex conglomeratus		r	f	f	f	f	21	1	2	1	E	144.77	134.12	150.33
Rumex crispus	f	o	f	f	f	f	21	1	2	1	E	169.03	181.74	150.87
Rumex hydrolapathum	f	r	r		r	f	43	5	2	3	P	192.64	189.22	180.28
Rumex maritimus			r	r	r	f	22	1	2	1	R	163.74	90.96	147.55
Rumex obtusifolius	f	o	f	o	o	o	17	3-5	1	3	E	164.71	196.59	152.75
Rumex scutatus		s	s				61	5	2	4	M	58.11	251.99	201.90
Rumex thyrsiflorus						o	51	1	2	3	R	225.33	0.00	203.50
Sagina procumbens		r	r	s			14	1	2	4	SC	80.33	233.98	161.52
Salvia pratensis	f		o			o	63	4	2	3	R	184.15	208.81	144.79
Sambucus ebulus		s	r				81	3	2	3	SC	38.74	236.85	174.37
Sanguisorba minor	f	s	o	r	s	s	63	4	2	4	R	170.36	232.98	128.07
Saponaria officinalis		o	f	r			16	4	2	2	P	76.34	226.93	147.86
Satureja vulgaris		r	r				83	4	2	3	M	58.11	251.99	201.90
Saxifraga granulata			f	s			51	3	2	2	P	33.84	197.64	95.44
Saxifraga tridactylites		r	r				62	5	2	4	M	58.11	251.99	201.90
Scabiosa columbaria	f	s	s				63	4	2	4	M	213.62	328.74	147.21
Scirpus maritimus			s	s	r	f	43	2	2	1	P	183.52	67.64	169.53
Scirpus sylvaticus	f	r	r	r	o	f	52	1	2	3	P	190.25	179.95	160.91
Scleranthus annuus					r		13	2	2	4	M	198.84	86.32	211.45
Scrophularia auriculata		s	f	r		r	43	1	2	1	P	100.58	160.84	129.86
Scrophularia umbrosa			r	r	s	r	43	1	2	1	SC	141.13	117.62	122.44
Scutellaria galericulata			o	o	o	f	43	2	2	3	P	154.26	104.96	138.94
Sedum acre		o	o	o	r	o	62	3	2	1	P	137.85	155.03	160.34
Sedum album		o	o	r	r		63	4	2	4	P	108.47	200.85	163.43
Sedum telephium		r	r	s			82	4	2	4	M	80.33	233.98	161.52
Sedum reflexum		r	s		s		62	5	2	4	M	107.82	221.93	224.94
Sedum sexangulare		r	o	s	r	s	62	5	2	1	R	113.85	172.12	172.59
Senecio aquaticus	o						52	1	1	1	R	291.38	367.11	119.86
Senecio erucifolius		r	f	r			51	1	2	3	SC	71.35	218.12	130.77
Senecio fluviatilis					r	o	44	1	2	1	P	214.73	34.53	206.68
Senecio jacobaea	f	f	f	f	f	f	62	3-5	2	4	E	166.83	186.56	156.44
Silaum silaus	f						51	1		3	E	291.38	367.11	119.86
Silene vulgaris		s	r				63	4	2	3	SC	38.74	236.85	174.37
Sinapis arvensis	f	f	f	f	f	f	11	1	1	1	E	166.83	186.56	156.44

Sisymbrium altissimum		r	r				16	4	2	1	SC	58.11	251.99	201.90
Sisymbrium officinale		r	f	r		o	15	3	2	3	P	113.35	158.63	150.61
Sisymbrium austriacum		o	f	r	r		16	1	2	1	SC	98.61	201.37	159.42
Solidago gigantea			s	r	s		44	1	2	3	R	134.31	154.19	82.69
Spergula arvensis				r	r		13	2	2	4	SS	184.02	124.13	105.72
Spergularia rubra			s	r	r		23	2	1	4	SS	147.21	140.61	108.44
Stachys palustris	f	f	f	f	f	f	44	1	2	3	E	166.83	186.56	156.44
Stellaria graminea		s	f	r	o		51	3	1	4	P	105.11	170.65	139.60
Stellaria media	r	r	r	r	r	f	11	1	2	1	E	175.18	159.91	163.16
Stellaria nemorum		s	s				91	3	2	3	R	58.11	251.99	201.90
Symphytum officinale	f	o	o	o	o	o	44	1	2	3	E	173.38	196.06	154.51
Tanacetum vulgare	r	o	f	r	o	r	17	3-5	2	3	E	144.81	189.72	163.24
Teesdalia nudicaulis					s		64	2	2	4	M	198.84	86.32	211.45
Thalictrum flavum	f	r	f	r	s		44	1	2	3	E	148.86	253.82	133.62
Thlaspi perfoliatum		s	r				12	4	2	1	SS	38.74	236.85	174.37
Thymus pulegioides	f	r	s	r	r	s	62	3	2	3	R	196.61	230.53	149.51
Thymus serpyllum					s		64	2	1	4	M	198.84	86.32	211.45
Torilis arvensis			s				51	3	2	3	SS	0.00	206.56	119.30
Torilis nodosa					r	o	51	1	2	3	P	214.73	34.53	206.68
Tragopogon pratensis	f	r	f	s		r	51	1	2	3	P	155.22	234.73	148.67
Trifolium arvense			r	s			64	2	2	4	SC	56.40	191.69	79.53
Trifolium campestre		s	r				62	5	2	4	R	38.74	236.85	174.37
Trifolium dubium	r	r	f	o	o	o	51	5	2	1	E	152.66	170.59	148.87
Trifolium fragiferum	f		r		r	o	21	1	2	3	P	203.56	186.74	159.22
Trifolium pratense	f	o	f	f	f	f	51	1	2	1	E	169.03	181.74	150.87
Trifolium repens	f	f	f	f	f	f	21	1	2	3	E	166.83	186.56	156.44
Trifolium striatum			s		s		62	1	2	1	M	99.42	146.44	165.37
Trisetum flavescens	f	r	f	f	f	f	51	1	2	3	E	171.43	176.48	144.79
Urtica dioica	f	f	f	f	f	f	82	5	2	1	E	166.83	186.56	156.44
Valeriana officinalis	f	r	f	r	o	f	52	1	2	3	P	170.22	182.75	156.53
Valerianella dentata		r	s				12	4	2	4	SC	77.48	267.14	229.43
Valerianella locusta	f	r	f	r			62	3-5	2	1	P	144.70	267.78	127.14
Verbascum nigrum		r	f		r		16	1	2	3	P	78.77	199.22	183.64
Verbascum phlomoides		r	r				16	4	2	4	M	58.11	251.99	201.90
Verbascum densiflorum		r	o				16	4	2	3	R	46.49	242.91	185.38
Verbascum thapsus		o	o	r			16	4	2	4	R	85.88	229.48	151.42
Verbena officinalis		o	f	s	s		51	3	2	3	SC	79.63	218.53	171.35
Veronica arvensis		r	o	r	s	r	62	3	2	4	R	122.03	162.47	154.54
Veronica chamaedrys	f	o	f	r			51	5	2	3	E	142.51	270.06	139.24
Veronica prostrata					s		63	4		4	M	198.84	86.32	211.45

Veronica scutellata	f						71	1	2	3	R	291.38	367.11	119.86
Veronica serpyllifolia	f	r	o	r	s	o	21	1	2	3	P	174.08	206.21	148.55
Veronica austriaca teucrium	f						63	4		4	P	291.38	367.11	119.86
Vicia sativa subsp. nigra		s	o		r		62	5	2	3	R	85.65	181.62	177.55
Vicia cracca	f	f	f	f	f	f	51	1	2	3	E	166.83	186.56	156.44
Vicia hirsuta		r	f	r			11	5	2	3	E	71.35	218.12	130.77
Vicia sativa subsp. sativa	f	r	f	o	o	f	51	3-5	2	3	E	170.17	181.71	148.70
Vicia sepium	f	f	f	f	f	f	82	3-5	2	3	E	166.83	186.56	156.44
Vicia tetrasperma subsp. gracilis			r				12	4	2	4	M	0.00	206.56	119.30
Vicia tetrasperma subsp. tetrasperma		r	o				11	4	2	4	R	46.49	242.91	185.38
Vulpia bromoides		o	f				64	4	2	1	SC	49.81	245.50	190.10
Rosa canina	o	o	o	r	r		84	5	1	3	R	150.68	239.21	153.38
Erigeron annuus			f	r			17	1	2	1	SC	56.40	191.69	79.53
Senecio inaequidens		f	f	f	f	f	44	1	2	1	E	141.92	150.45	163.75
Xanthium strumarium	f	r	o	r	r	o	15	1	2	1	P	175.63	198.72	152.48

Maasvakken: I: Lorraine Meuse, II: Ardennes Meuse, III: Common Meuse, IV: Peelhorst Meuse, V: Sand Meuse, VI: Bergse Meuse.

* OECOEGRP: Ecological Groups sensu Stieperaere & Fransen, 1982.

† Ecoregions: 1: fluvial region, 2: Campine region, 3: Loamy region, 4: Calcarous region, 5: Vosges-Ardennes-Eifel region.

_ Disturbance adaptations: 1: invader, 2: endurer, 3: resister, 4: avoider.

§ Population dynamics: M: metapopulation, SS: source-sink, R: remnant, SC: shifting-cloud, P: patchy, E: extended local.

Appendix table S2:

List of the surveyed species (n= 209). 1: typical river species; 2: typical forest species; 3: woody species; 4: highest frequency in summer bed (S), winter bed (W) or disconnected (D) sites. Species are ranked ascending according to their DCA scores on the first axis, i.e. with increasing tolerance against flooding.

Carex sylvatica		x		D
Ribes nigrum	x	x	x	D
Polygonatum multiflorum		x		D
Viola riviniana		x		W
Hedera helix		x		D
Ranunculus ficaria		x		D
Veronica hederifolia		x		W
Ulmus minor	x		x	D
Arum italicum				D
Corylus avellana		x	x	D
Milium effusum		x		D
Tilia sp.			x	D
Ornithogalum umbellatum		x		D
Ribes rubrum		x	x	W
Acer pseudoplatanus			x	D
Evonymus europaeus	x	x	x	DW
Circaea lutetiana		x		D
Ulmus glabra			x	W
Sorbus aucuparia		x	x	D
Lamium maculatum	x			W
Geranium robertianum				D
Solanum dulcamara				S
Vaccinium vitis-idaea			x	D
Iris pseudacorus				D
Carex riparia				W
Crepis capillaris				W
Senecio jacobaea				W
Glechoma hederacea				DW
Sonchus arvensis				D
Populus tremula			x	D
Parietaria officinalis		x		DW
Poa nemoralis		x		D
Angelica sylvestris				D
Myosotis palustris				W
Malus sylvestris		x	x	W
Mentha suaveolens	x			W
Carex acutiformis				W
Populus nigra	x		x	D
Lycopus europaeus				S
Cardamine hirsuta				W
Filipendula ulmaria				W
Crataegus laevigata	x	x	x	W
Crepis biennis	x	x		W
Stellaria holostea		x		D
Alopecurus pratensis				D
Cornus sanguinea	x	x	x	D
Eupatorium cannabinum				W
Epipactis helleborine		x		W

	1	2	3	4
Cardamine amara	x	x		W
Impatiens parviflora		x		W
Stellaria nemorum	x	x		D
Stachys sylvatica		x		D
Brachypodium sylvaticum		x		D
Caltha palustris				W
Athyrium filix-femina		x		D
Ligustrum vulgare		x	x	W
Thelypteris palustris				D
Mycelis muralis	x	x		W
Sambucus nigra		x	x	D
Impatiens glandulifera	x			W
Viburnum opulus		x	x	D
Juglans nigra			x	W
Alliaria petiolata				W
Alnus glutinosa		x	x	W
Carex remota		x		D
Fraxinus excelsior			x	W
Aegopodium podagraria				W
Calystegia sepium				W
Salix triandra	x		x	S
Salix x mollissima			x	S
Arctium lappa				W
Salix viminalis	x		x	S
Convolvulus arvensis				W
Primula veris				W
Galeopsis tetrahit				W
Potentilla anserina				D
Campanula rapunculus				W
Juncus compressus				W
Plantago major				W
Salix x rubens			x	S
Clematis vitalba				W
Epilobium hirsutum				W
Galium aparine				D
Polygonum amphibium				D
Mentha aquatica				W
Quercus rubra			x	D
Festuca arundinacea				S
Thalictrum flavum	x			W
Prunus spinosa		x	x	D
Agrostis canina				W
Viola odorata		x		W
Rubus caesius		x	x	W
Poa pratensis				D
Vicia sp.				D
Juncus effusus				D
Scrophularia nodosa		x		W
Salix caprea		x	x	W
Epilobium tetragonum				D/W
Phalaris arundinacea	x			S
Ranunculus acris				D
Rubus fruticosus agg.		x	x	W
Senecio inaequidens	x			W
Salix purpurea	x		x	S
Sinapis arvensis				S
Salix cinerea		x	x	D
Helianthus tuberosus	x			S
Scrophularia auriculata	x			W
Stachys palustris				D
Elymus repens				D
Prunella vulgaris				W
Carex riparia				WD
Chenopodium album				W
Rumex crispus				W
Euphorbia esula	x			S
Geranium rotundifolium	x			W
Barbarea vulgaris				W
Juncus inflexus				W
Artemisia vulgaris				S
Lolium perenne				W

	1	2	3	4
Cardamine impatiens		x		W
Myosotis sylvatica				D
Rosa rubiginosa	x		x	W
Veronica officinalis				W
Urtica dioica				W
Dryopteris filix-mas			x	D
Salix aurita			x	W
Poa palustris				W
Lythrum salicaria				W
Ranunculus repens				W
Carex vesicaria				W
Eleocharis palustris				W
Rorippa amphibia				W
Chaerophyllum temulum				W
Symphytum officinale				D
Silene dioica				W
Salix alba	x			W
Cardamine flexuosa			x	W
Sisymbrium altissimum				W
Euphorbia esula ssp. esula		x		S
Brassica nigra				W
Silene vulgaris				W
Verbascum nigrum	x			W
Lactuca serriola				W
Carex hirta				W
Heracleum sphondylium				W
Stellaria media				D
Geranium molle				W
Quercus robur			x	D
Matricaria maritima				W
Saponaria officinalis		x		W
Dactylis glomerata				W
Rorippa sylvestris				W
Ajuga reptans		x		D
Capsella bursa-pastoris				W
Reseda lutea	x			W
Sisymbrium officinale				W
Stellaria aquatica				W
Melilotus altissima		x		W
Bryonia cretica ssp. dioica		x		W
Tanacetum vulgare				S
Chelidonium majus				W
Torilis japonica		x		W
Potentilla reptans				W
Crataegus monogyna		x	x	D
Cardamine pratensis				W
Vicia cracca				W
Cruciata laevipes		x		W
Lathyrus pratensis				W
Prunus avium		x	x	D
Anthriscus sylvestris				W
Cirsium arvense				W
Galium mollugo	x			W
Stellaria graminea				W
Deschampsia cespitosa			x	W
Vicia sepium		x		D
Pimpinella major				D
Geranium dissectum				W
Lotus corniculatus				W
Poa trivialis				D
Agrimonia eupatoria			x	D
Arrhenatherum elatius				W
Carex spicata				W
Veronica chamaedrys				W
Holcus lanatus				D/W
Hypericum perforatum				W
Prunus serotina			x	D
Rosa canina		x	x	D
Plantago lanceolata				W
Equisetum arvense				W
Cerastium fontanum ssp. vulgare				W
Origanum vulgare				W

Appendix table S3:

Rare floodplain meadow species (< 5 plots), with DCA-axis values and Ellenberg indicator values.

Species	dca1	dca2	dca3	dca4	Grwtr-ind	Light-ind	T-ind	Alc-ind	N-ind
Agrostis canina	3.15	2.7305	1.35	3.2197	4	9	5	3	2
Hordeum murinum	6.2098	2.7054	-0.557	0.5255	7	8	7	7	5
Poa compressa *	6.3509	3.7183	-0.5473	3.297	7	9	X	9	3
Sedum reflexum *	6.9255	1.4549	-0.8738	1.6271	7	7	5	5	1
Phleum pratense	4.5862	3.4718	1.1831	1.6532	7	7	X	X	6
Sedum sexangulare *	6.2987	3.9804	1.1424	3.2984	7	7	5	6	1
Alopecurus geniculatus	-0.1756	1.9911	3.4257	0.2342	4	9	6	7	7
Myosotis ramosissima	5.8397	4.3012	6.4063	-0.3278	7	9	6	7	1
Vicia sativa ssp. nigra *	5.6335	2.9009	5.6738	-0.2482	7	5	6	X	X
Rhinanthus minor *	6.4232	-0.025	4.1809	-0.1997	5	7	5	X	3
Primula veris *	6.1999	-2.6253	3.3509	3.9412	7	7	X	8	3
Cerastium arvense	6.0884	2.0325	3.5244	-0.8431	7	8	X	6	4
Silene latifolia (subsp. alba) *	5.955	1.144	2.6546	-0.9788	7	8	6	X	7
Plantago media *	5.9031	2.5815	3.8824	4.6739	7	7	X	7	3
Valerianella locusta *	5.4285	3.0358	4.9438	-0.7906	7	7	6	7	6
Aphanes inexpectata	7.1017	2.5747	6.1793	1.5724	7	7	7	4	4
Erophila verna	6.8217	4.0832	5.8389	-0.5451	7	8	6	X	2
Galeopsis angustifolia *	5.6772	2.8352	1.1258	3.4504	7	8	7	8	4
Potentilla argentea *	6.071	2.9253	1.4245	3.0585	7	9	6	3	1
Veronica serpyllifolia	4.0688	1.5084	1.7351	3.7625	9	X	X	5	X
Erodium cicutarium	5.8299	2.868	6.1864	-0.0976	7	8	6	X	X
Tragopogon pratensis *	5.1815	1.6891	1.7524	1.3083	7	7	6	7	6
Ononis repens ssp. Spinosa *	5.2813	3.4365	1.9378	3.7338	7	8	6	7	3
Rumex acetosella	6.1172	1.4938	4.9323	0.0949	7	8	5	2	2
Eryngium campestre *	6.0502	2.9476	1.6798	3.6683	7	9	7	8	3
Cruciata laevipes *	3.847	0.2644	3.529	2.4695	7	7	5	6	7
Sedum telephium *	5.4582	0.7516	2.532	1.9067	7	7	6	7	X
Salvia pratensis *	5.4757	1.6609	1.1361	0.1903	7	8	6	8	4
Avenula pratensis *	4.1856	1.8801	4.8539	2.6574	7	7	6	X	2

Silene vulgaris	*	4.2467	3.443	4.3888	-0.821	7	8	X	7	2
Malva moschata	*	4.6198	2.9725	5.2329	0.9383	7	8	6	7	4
Deschampsia cespitosa		2.889	1.2157	4.0621	2.161	4	6	X	X	3
Vulpia myuros	*	5.5679	4.5686	2.6525	1.9198	7	8	7	5	1
Erigeron annuus		4.3518	4.5141	2.6147	-0.0596	7	7	6	X	8
Campanula rotundifolia		6.796	3.917	5.3004	5.1466	7	7	5	X	2
Saxifraga tridactylites	*	7.5234	5.088	6.5733	-1.1146	7	8	6	7	1
Trifolium campestre	*	5.3446	3.8496	3.2629	1.3446	7	8	6	6	3
Poa palustris		1.5747	0.6598	5.2529	3.8773	3	7	5	8	7
Geranium pyrenaicum	*	4.9261	3.585	5.2459	-0.3723	7	8	6	7	8
Torilis arvensis	*	4.7953	4.4106	3.6343	0.015	7	7	7	9	4
Hieracium pilosella		6.614	3.3318	5.7459	4.5434	7	7	X	X	2
Cerastium semidecandrum		6.0819	5.6103	3.2287	3.3459	7	8	6	6	X
Dianthus armeria	*	5.0906	4.5983	5.5855	0.9875	7	6	6	X	3
Vicia tetrasperma	*	5.5735	3.3724	5.4733	-0.0696	7	6	6	5	5
Sagina apetala		6.6408	5.7113	5.8391	-1.1071	7	8	7	4	4
Arabidopsis thaliana		7.1411	5.4676	6.1795	-2.2359	7	6	6	4	4
Italics: diagnostic species of dry river grassland communities, *: river corridor species										

Appendix Table S4.

Rare species (river corridor species with *): ordination axis scores, appointed population dynamic strategies, recruitment analysis result

Species		dca1	dca2	dca3	population strategy	[illegible] colonization
Agrostis canina		3.15	2.7305	1.33	P	2
Hordeum murinum		6.2898	2.7184	0.587	P	2
Poa compressa	*	6.5505	3.7165	0.5475	P	10
Sedum reflexum	*	6.5255	1.4540	0.8738	M	2
Phleum pratense		4.5662	3.4716	1.1831	P	5
Sedum sexangulare	*	6.2587	3.9864	1.1424	P	7
Alopecurus geniculatus		0.1738	1.9811	9.4237	R	[illegible]
Myosotis ramosissima		3.8097	4.3012	6.4065	P	4
Vicia sativa nigra	*	5.6335	2.5800	8.6736	P	3
Rhinanthus minor	*	6.4232	0.025	4.1800	P	2
Primula veris	*	6.1959	2.6255	9.8808	P	3
Cerastium arvense		6.0884	2.0325	9.3244	P	2
Silene latifolia alba	*	3.735	1.144	2.6846	P	2
Plantago media	*	5.9131	-2.9815	5.8824	R	1
Valerianella locusta	*	5.4285	3.0736	4.9438	M	9
Aphanes inexspectata		7.1117	2.4547	6.1759	[illegible]	4
Erophila verna		6.8217	4.6672	8.8081	P	4
Galeopsis ladanum angustifolia*		5.6772	2.8332	1.1250	M	1
Potentilla argentea	*	6.171	2.9258	1.4245	R	0
Veronica serpyllifolia		4.0888	1.5084	1.7381	P	8
Erodium cicutarium		8.0239	-2.808	6.1864	P	8
Tragopogon pratensis	*	5.1815	1.6851	1.7324	R	5
Ononis repens spinosa	*	5.2813	3.4365	1.0376	R	4
Bromus racemosus		6.1172	1.4038	4.9423	M	6
Eryngium campestre	*	6.0302	2.9476	1.6709	R	2
Cruciata laevipes	*	3.847	0.2644	3.321	R	4
Sedum telephium	*	5.4382	0.7301	2.532	R	0
Salvia pratensis	*	5.4737	1.4648	1.1361	P	4
Avenula pratensis	*	4.1836	1.0803	4.8539	R	1
Silene vulgaris	*	4.2417	3.443	3.0068	P	3
Malva moschata	*	4.4186	2.0725	9.2321	P	2
Deschampsia cespitosa		2.889	1.2137	3.0823	R	0
Vulpia myuros	*	5.3670	3.3686	2.6525	P	11
Leucojum aestivum		6.3318	4.5143	2.6147	[illegible]	8
Campanula rotundifolia		6.756	3.917	9.8504	P	5
Saxifraga tridactylites	*	7.5254	3.088	6.5738	M	4
Trifolium campestre	*	5.5446	1.8250	9.2629	P	8
Poa palustris		1.5747	0.6598	9.2329	P	0
Geranium pyrenaicum	*	6.5264	3.865	8.2430	P	2
Torilis arvensis	*	4.7953	4.4106	9.9348	M	1
Hieracium pilosella		6.614	8.3318	9.7450	M	0
Cerastium semidecandrum		6.5810	5.6105	9.2287	P	8
Dianthus armeria	*	5.5906	2.5080	5.9455	M	3
Vicia tetrasperma tetrasperma	*	8.5738	3.3724	5.4588	P	8
Sagina apetala		6.6408	8.7113	8.8091	P	4
Arabidopsis thaliana		7.1411	5.4670	6.1795	P	4
Medicago falcata	*				P	4
Rhinanthus alectorolophus	*				M	1
Briza media	*				R	6
Carex caryophyllea	*				R	0
Thymus pulegioides	*				M	0
Geranium pratense	*				M	2
Potentilla neumanniana	*				R	0

Appendix tables S5:

Mapping survey results for the studied reaches of the Allier, Loire, Dordogne and Meuse.

Appendix table S5.1. Landscape (land use categories) and riparian forests in the mapping of the four reaches

		Allier %	Loire %	Dordogne %	Meuse %
River hydrosystem	running water	9,3	7,1	14.3	13.1
	ponds, oxbow lakes	1,0	1,2	0.3	0.8
Natural floodplain	sand-gravel bars	9,3	4,7	3.8	0.1
	pioneer vegetation	2,2	0,2	0.2	0.1
	Shrubland	3,9	5,0	2.7	3.2
	Salicetum triandrae-viminalis	1,7	0,3	0.1	0.2
	Salicetum purpurea	0,7	0,3	1.0	0.1
	Salicetum albo-fragilis	1,7	0,0	1.4	3.8
	Salici-populetum	4,2	3,0	1.4	0
	Populetum	5,3	3,5	3.2	0
	hardwood forest	2,4	5,1	2.7	4.9
Agricultural use	floodplain meadows	19,8	37,9	0.9	3.0
	Pastures	6,8	9,6	16.4	16.8
	arable land	28,0	19,3	44.4	40.0
	plantations	2,1	1,4	3.9	0.6
Artificial/anthropic units	infrastructure/urbanisation	1,9	1,2	2.5	2.7
	gravel pits	0	0	0.8	10.9

Appendix table S5.2. Forest community patches and area in the four reaches

	Loire Lamenay-Gannay-s-Loire 32km		Allier Ch_tel-de-Neuvre-Moulins 32km		Dordogne Souillac-Vitrac 25km		Maas Grensmaas-Maaseik 35km	
	# patches	ha	#	ha	#	ha	#	ha
Salix purpurea thicket	35.0	10.0	83.0	22.0	25.0	14.0	6.0	1.2
Willow thicket	76.0	13.0	72.0	56.0	2.0	1.2	4.0	3.9
Total thickets	111.0	23.0	155.0	78.0	27.0	15.2	10.0	5.1
Thicket/riverkm	3.5	0.7	4.8	2.4	1.1	0.6	0.3	0.1
Salici-Populetum	157.0	119.0	257.0	139.0	20.0	20.2	0.0	0.0
Salicetum albo-fragilis	12.0	1.4	104.0	54.0	13.0	20.3	22.0	76.0
Dry Populetum	384.0	189.0	165.0	173.0	11.0	47.3	0.0	0.0
Total Riparian communities	441	279.9	681	444	71	103	32	91.1
Ripfor/km	13.8	8.8	21.3	13.87	2.8	4.1	0.9	2.3
Hardwood forest	349.0	149.0	105.0	78.0	5.0	40.2	16.0	56.0
Total developed forest	882.0	456.4	631.0	444.0	49.0	128.0	38.0	134.0
Forest/km	27.6	14.3	19.7	13.9	2.0	5.1	1.1	3.8

Appendix S5.3. Forest development measures for the river reaches. The area and frequency measures are calculated as area and frequency of riparian forest per river kilometre stretch.

Measures	Allier	Loire	Dordogne	Maas
Patch area	13,9	8,71	4,12	2,3
Patch area thicket	2,4	0,7	0,6	0,1
Patch frequency	21,3	13,8	2,8	0,9
Patch frequency thicket	4,8	3,5	1,1	0,3
Perimeter/area	12,8	13,2	8,7	0,4
Perim/area thicket	7,8	5,1	5,6	1,1

Curriculum Vitae

Kris Van Looy werd op 23 september1970 geboren te Antwerpen, waar hij tevens opgroeide en school liep op de linkeroever van de Schelde. In 1993 behaalde hij de graad van bioingenieur, specialisatie Ruimtelijke Ordening en Milieu, aan de Katholieke Universiteit Leuven. Eerste werkervaringen werden opgedaan op het Instituut voor Land- en Waterbeheer (KUL), aan het Ruimtelijk Structuurplan Vlaanderen, deelfacet open ruimte, en op de Afdeling Natuur van AMINAL, aan de afbakening van de Groene Hoofdstructuur. Vervolgens mocht hij voor het Instituut voor Natuurbehoud vanuit de toenmalige vestiging in Kiewit bij Hasselt terug het Walenbos in, waar hij zijn eindwerk mocht vervolledigen tot een ontwerpbeheersplan. Aansluitend werd hij op ontdekking gezonden naar de Limburgse Maasvallei, voor de ecologische advisering van het Maasdijkenplan en de ontwikkeling van een ruimtelijke visie in aansluiting bij de natuur- en hoogwaterbeschermingsplannen aan Nederlandse zijde. Ondertussen is hij reeds 11 jaar betrokken bij de uittekening van het rivierherstelproject voor de Grensmaas, met het ontwerpen van de ingrepen, de studie van de herstelpotenties en van de natuur langs de rivier in z'n geheel. In 2001 in het huwelijk getreden met Anouck Jonckheere en zij werden verblijd met de geboortes van twee zonen Siemen (2002) en Wannes (2004) en zijn in blijde verwachting van een derde.

Wetenschappelijke
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wijze van citeren/how to cite

Van Looy K. [2006]. River Restoration & Biodiversity Conservation. A disorder approach. Research Institute for Nature and Forest, Brussels. 392 pp.

Van Looy K. & Patrick Meire [2006]. A conservation paradox for river corridor. In: Van Looy K.[2006]. River Restoration & Biodiversity Conservation. A disorder approach. Research Institute for Nature and Forest, Brussels. pp. 175-193.

Central question for the research was the role played by river dynamic processes in the river ecosystem and its regulatory aspects, useful for the development of conservation and restoration strategies. These aspects were investigated in the terrestrial riverine communities of floodplain vegetation and riparian ground beetles and forests. This thesis contains a number of papers featuring a range of river restoration and biodiversity conservation topics, brought in the picture at different scales with an array of techniques and approaches for a wide variety of biotic communities, emphasized upon in habitat templets, population dynamic strategies, habitat networks or diagnostic species. Yet, they all tell the same story of a river expressing itself in its unique setting of geomorphology, landscape and biotic features, in a non-equilibrium relation that is governed by the flow dynamics. These observations were integrated in a river disorder approach for the target setting and objective definition of the restoration and conservation strategies. Guidelines and targets were derived for local, reach or even catchment's scale conservation strategies, based on determined responses to disorder elements of specific communities.