A Landscape Approach towards Ecological Integrity of Catchments and Streams

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Abstract

Landscape ecology principles were applied to study how in-stream structures as well as terrestrial land cover and land use affect the ecological integrity of streams. Focusing on the role of habitat factors at multiple spatial scales of catchments as independent variables, macroinvertebrates, brown trout (*Salmo trutta*) and freshwater pearl mussels (*Margaritifera margaritifera*) were used as dependent response variables. Special efforts were made to capture the range of impacts along the gradient from altered to near-natural catchments in Europe's centre and north. With 25 catchments in the Carpathian Mountains in Central Europe as a landscape laboratory and a natural experiment design I studied the relationships between catchment land cover composition, riparian vegetation, and in-stream habitat characteristics on the one hand, and benthic macroinvertebrate assemblages on the other. The most important variables were at the terrestrial catchment level. The usefulness of data representing higher taxonomic levels and the use of the abundance of individuals from the orders Ephemeroptera, Plecoptera and Trichoptera, as surrogates for species richness was evaluated. Plecoptera was identified as the most effective group. Comparison of the abundance and taxonomic richness of the Plecoptera order in relation to catchment forest cover and water chemistry demonstrated the occurrence of non-linear response to forest cover. Using Swedish data I showed that it is possible to predict viable freshwater pearl mussel populations from riparian land cover, water chemistry and the abundance of the host fish species (brown trout). In another study a positive effect of woody debris on the abundance and size of brown trout in Swedish forest streams was observed. Altogether these results stress the importance of adopting a multiple scale perspective when assessing ecological integrity in riverine landscapes, from the amount of local in-stream structures as large woody debris (LWD) to the effects of riparian and catchment land cover composition and land use. To support implementation of EU policies about good ecological status a systematic landscape approach including (1) quantification of the ecological integrity concept, and (2) collaborative and communicative bottom-up participatory approaches to spatial planning need to be combined.

Keywords: landscape ecology, ecological integrity, riverine landscapes, riparian zones, large woody debris, ecological status, policy implementation, spatial planning.

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Dedication

This thesis I dedicate to those we live with and love and should try to know better before they elude us…

"I sat there and forgot and forgot, until what remained was the river that went by and I who watched. On the river the heat mirages danced with each other and then they danced through each other and then they joined hands and danced around each other. Eventually the watcher joined the river, and there was only one of us. I believe it was the river.

Even the anatomy of a river was laid bare. Not far down-stream was a dry channel where the river had run once, and part of the way to know a thing is through its death. But years ago I had known the river when it flowed through this now dry channel, so I could enliven its stony remains with the waters of memory.

In death it had its pattern, and we can only hope for as much. Its overall pattern was the favourite serpentine curve of the artist sketched on the valley from my hill to the last hill I could see on the other side. But internally it was made of sharp angles. It ran seemingly straight for a while, turned abruptly, then ran smoothly again, then met another obstacle, again was turned sharply and again ran smoothly. Straight lines couldn't be exactly straight and angles that couldn't have been exactly right angles became the artist's most beautiful curve and swept from here across the valley to where it could be no longer seen."

Norman Maclean
A river runs through it

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List of Publications

This thesis is based on the work contained in the following papers, referred to by Roman numerals in the text:

I Törnblom, J., Angelstam, P., Degerman, E., Henrikson, L., Edman, T., Temnerud, J. 2007. The relative roles of landscape, riparian and instream factors to macroinvertebrate composition in local catchments: changed country borders as landscape experiments (submitted manuscript).

II Törnblom, J., Roberge, J-M., Angelstam, P., Rapid assessment of macroinvertebrate species richness in second and third order streams – an evaluation of surrogates across a gradient of land-use intensity in the Carpathian Mountains (manuscript).

III Törnblom, J., Degerman, E. Angelstam, P. Dose-response relationship using the benthic macroinvertebrate Plecoptera as a bioindicator for ecological integrity of catchments (manuscript).

IV Törnblom, J., Degerman, E. Angelstam, P., Henrikson, L., Söderberg, H., Norrgrann, O., Andersson, K. Predicting occurrence of viable populations of freshwater pearl mussels, *Margaritifera margaritifera* (L.), in Swedish boreal forest (manuscript).

V Degerman, E., Sers, B., Törnblom, J., Angelstam, P. (2004). Large woody debris and brown trout in small forest streams – towards targets assessment and management of riparian landscapes. Ecological Bulletins 51:233-239.

VI Törnblom, J., Angelstam, P. Implementing the EU Water Framework Directive: Towards a hierarchical toolbox for achieving Good Ecological Status in riverine landscapes (manuscript).

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Abbreviations

CCA	Canonical Correspondence Analysis
EPT	Ephemeroptera, Plecoptera, Trichoptera
FPM	Freshwater Pearl Mussel
GIS	Geographical Information System
HSI	Habitat Suitability Index
IUCN	International Union for Conservation of Nature
LWD	Large Woody Debris
PCA	Principal Component Analysis
SEPA	Swedish Environmental Protection Agency
SERS	Swedish Electrofishing RegiSter
TAGA	TerrAquatic Gap Analysis
WCED	World Commission on Environment and Development
WFD	Water Framework Directive

1 Introduction

Gap analysis is a tool for strategic level assessment of the extent to which environmental policies succeed in maintaining biodiversity by protection, management and restoration of habitat networks (Scott *et al.*, 1993, 1996). Originally developed in the USA, gap analyses have been used in terrestrial systems to increase society's awareness about conservation needs, and to guide the practical implementation of such policies (Scott *et al.*, 1993; Sowa, 1999; Angelstam & Andersson, 2001; Lõhmus *et al.*, 2004; Scott & Schipper, 2006). The rationale for focusing on the functionality of habitat networks is that they serve as proxies for the maintenance of viable populations of species, and vital ecosystem processes and ecological integrity.

Originally gap analyses focused on representation, i.e. that the different types of conservation areas should represent the natural composition of different ecosystems in an ecoregion (Margules & Pressey, 2000). Angelstam & Andersson (2001) developed the idea further by combining quantification of the habitat area with information about thresholds for the amount and quality of habitats needed to maintain viable populations within an ecoregion. This approach is now the basis for both the Swedish forest protection strategy (Angelstam & Andersson, 2001; SUS, 2001), and Swedish state forest company Sveaskog's quantitative biodiversity maintenance goals (Angelstam *et al.*, 2006). Quantitative regional gap analyses have also been applied recently for Estonia (Lõhmus *et al.*, 2003) and Latvia (Angelstam *et al.*, 2006). A regional quantitative gap analysis contains the following steps: (1) measure today's amount of different representative habitats in an ecoregion, (2) compare the amounts with ecologically-based performance targets interpreting the relevant policies (such as maintaining viable populations of naturally occurring species), (3) conclude whether there is a gap or not, and if so for what habitat, and where. Analyses can be made at multiple scales, ranging from a country to a land management unit or a watershed.

For aquatic ecosystems in riverine landscapes there is no such tradition in planning of systematic analyses for conservation and restoration management (Wiens, 2002). There is, however, a growing insight that there are complex interactions between the terrestrial and aquatic systems on the one hand, and social systems involved with policy implementation by governance, planning and management on the other. Being a social-ecological system the term landscape approach (*e.g.*, Singer, 2007) captures this need for applied interdisciplinary, i.e. transdisciplinary (Tress & Tress, 2003; Angelstam *et al.*, 2004; Tress *et al.*, 2005), approaches. The term landscape approach also emphasizes broad spatial scales and the ecological effects of the spatial patterning of ecosystems. Specifically, it considers (a) the development and dynamics of spatial heterogeneity, (b) interactions and exchanges across heterogeneous landscapes, (c) the influences of spatial heterogeneity on biotic and abiotic processes, and (d) the management spatial heterogeneity (Risser *et al.*, 1984). Angelstam *et al.* (2004) reflects the idea that landscapes evolve through time, as a result of being acted upon by natural forces and human beings, which underlines that landscapes forms a whole, whose natural and socio-cultural components are taken together, not separately (Berkes *et al.*, 2003).

Land managers have traditionally assumed that achieving maximum local habitat diversity will favour diversity of wildlife. Recent trends in species composition in fragmented landscapes suggest, however, that a more comprehensive view is required for perpetuation of regional diversity (Noss, 1983). A regional network of reserves, with sensitive habitats insulated from human disturbance, might best perpetuate ecosystem integrity in the long term (Noss, 1983). Ecological integrity has been expressed as the maintenance of all internal and external community processes and attributes so that high ecological integrity corresponds to a natural state and where the natural community is preserved by regulation, resilience, and resistance to environmental stress (Moog, 1995; Moog & Chovanec, 2000). This definition is analogous to Karr's (1990) definition of ecological (or biological) integrity. A major goal of conservation is the perpetuation of indigenous ecosystem's structure, function, and integrity. Thus parks, reserves, and wildlife areas should have perpetuation of natural ecosystems as a principal goal (Noss, 1983).

A landscape can be perceived as a contiguous area, intermediate in size between an ecoregion and a site, with a specific set of economic, ecological, socio-cultural characteristics. In addition a landscape can also be perceived as a social system with institutions and people representing different actors and stakeholders in public, private and civil sectors at multiple levels. This is consistent with the idea that landscapes are social-ecological systems (Norton, 2003; Berkes *et al.*, 2003). While forest managers try to accommodate commodity and non-commodity values in the same management unit, conservationists often define functional conservation landscapes, and other stakeholders such as farming communities may define their cultural or livelihood landscapes (*e.g.*, Innes & Hoen, 2005). But individual ecosystems, the traditional focus of ecology, should not be seen as separate entities (Hansson, 1977). Almost all ecosystems are "open" and exchange energy, mineral nutrients, and species. Particularly in highly heterogeneous regions, the landscape mosaic may be a more appropriate unit of study and management than single sites or ecosystems. Landscape has been variously defined, usually in somewhat ambiguous terms. Forman and Godron (1981) suggest a more precise definition of landscape as a "kilometres-wide area where a cluster of interacting stands or ecosystems is repeated in similar form." A landscape is therefore an ecological unit with a distinguishable structure. The importance of the landscape concept is in its recognition that the structural components of a landscape interact (Forman, 1981; Forman & Godron, 1981). This is consistent with the view that the functionality of protected area networks is dependent on the quality of the matrix surrounding them.

The EU Water Framework Directive (Directive, 2000/60/EC) (WFD) has recently implicitly reinforced the application of an applied interdisciplinary, or transdisciplinary, approach. Inspired by the successful implementation of landscape ecological principles in terrestrial systems, Wiens (2002) and Rabeni & Sowa (2002) forcefully argued that such principles should also be applied when analysing riverine landscapes. The ecological principles include the importance of: (1) variable patch quality, (2) patch boundaries affecting flows, (3) patch context, (4) connectivity, (5) organisms and (6) scale. Improving the performance of natural resource governance and management within specific social-ecological systems, or simply landscapes and catchments, requires an understanding from politicians, civil servants and stakeholders in the context of ecological integrity of temporal and spatial boundaries of the system itself. The policies associated to the implementation of the 16 Swedish environmental quality objectives, the WFD's ambition to reach good ecological water status, or the Swedish forestry acts' ambition to preserve all naturally occurring species in viable populations, expect the fulfilment of these documents declarations. In addition issues perceived as concerns by the actors and stakeholders in the social-ecological system, *i.e.* landscape, and policy objectives, institutions, instruments and organizations addressing these issues must be understood in order to implement and concretize expected functionality. It is possible to improve the policy implementation by governance and management of ecosystems by systematically identifying institutional contexts at play, i.e. people, agencies, and policy processes (Lee, 1993; Clark *et al.*, 1996). In order to protect, manage and restore ecological integrity it thus becomes necessary to identify the institutional barriers, which also may include major social and economic forces that are currently driving the loss of functional diversity and to create incentives to redirect those forces (Folke *et al.*, 1996; Sabatier *et al.*, 2005b; Pahl-Wastl, 2006).

Because aquatic systems have been viewed as discrete systems in general, separate from their surrounding landscapes and therefore easy to collect data in (Schneider *et al.*, 2002), there is a very long tradition of collecting data of chemical, physical and biological variables. Good examples are pH, conductivity, alkalinity, dissolved organic carbon (DOC), light, erosion, stream morphology, temperature, fish and macroinvertebrates. For systematic analyses of riverine landscapes analogous to terrestrial gap analyses there is, however, a need to evaluate different indicator variables and develop performance targets that define ecological integrity. Relationships between woody debris in water, and fish and benthos (Sundbaum & Näslund, 1998; Lemly & Hildebrand, 2000; Zalewski, 2002; Degerman *et al.*, 2004), sediment load and fish (Eriksson & Nyberg, 2001; Richardson & Lowett, 2002), insolation and benthos (Olsson, 1995), suggest that the indicator approach is indeed feasible. However, along with performance targets and new insights concerning thresholds for "how much is enough" of habitats within the different spatial scales of a catchment, it is also necessary to understand the processes that affect the composition and structure of ecosystems (Noss, 1990; Angelstam & Kuuluvainen, 2004).

The ecological consequences of land cover and land use change for biodiversity and ecological integrity of rivers and streams is a prominent environmental issue worldwide (Karr & Chu, 1999; Moore & Palmer, 2005). The decline of species richness, abundance and diversity, along with the loss of ecosystem goods and services provided by river systems with intact ecological integrity (Costanza, 1991), are global problems (Dynesius & Nilsson 1994; Master *et al.*, 1997; Naiman & Turner, 2000; Nilsson *et al.*, 2005). The forces driving degradation of rivers and streams include anthropogenic fragmentation and loss of habitat, pollution, and introduction of exotic species. Achieving ecological integrity thus usually requires active ecosystem restoration (Karr, 1991; Karr & Chu, 1999). The concept of ecological integrity was developed to capture the complex changes in ecosystem under anthropogenic stress (Frey, 1977; Karr, 1991). True ecological integrity is supposed to occur in pristine systems without human influence. Areas characterised by such reference conditions for biological integrity are assumed to support a community composition that is the product of evolutionary and biogeographical patterns and processes (Karr, 1991; Angermeier & Karr, 1994; Karr & Chu, 1999). To implement policy ambitions about ecological integrity necessitates new and innovative approaches to adaptive management and governance as well as assessment that are applied across multiple spatial and temporal scales, from in-stream habitats to entire catchments and regions.

The biodiversity concept encompasses compositional, structural and functional elements at multiple spatial scales (Noss, 1990). For forest ecosystems Larsson *et al* (2001) used natural forest dynamics as a benchmark for defining what these elements are. The same approach is suggested for rivers and streams in riverine landscapes in this thesis. The geomorphology of the valley determines the soil and availability of ions, and also the slope of the land. Soil, water and climate shape the vegetation. In turn, the vegetation determines the supply of organic matter together with the soil that influences water chemistry and water inputs to the stream (Hynes, 1975; Minshall *et al.*, 1985; Naiman *et al.*, 1987; Moore *et al.*, 1991; Chipman & Johnson, 2002).

In managed landscapes human activity cause large and cumulative effects on catchments, streams and rivers, sometimes direct and obvious, sometimes more subtle (Giller & Malmqvist, 1998). This means that rather than simply associated with the stream channel it self, management of stream ecosystems should include entire catchments from multiple perspectives. In order to understand the level of ecological integrity of streams and rivers holistically, it is thus necessary to consider the entire drainage basin, incorporating both the aquatic system and its surrounding catchment from micro level to macro landscape levels (Hynes, 1975; Richards *et al.*, 1997; Townsend *et al.*, 1997). In addition EU and the Swedish Parliament explicitly, through associated directives and policies, expect viable populations of all naturally occurring species, good ecological water status as well as sustainable development. However, the concept of sustainable development in Sweden has changed from an ecological perspective to a more general focus on economical and social perspectives. This means that the connection between sustainable development and environmental politics has become weaker (Lundqvist & Carlsson, 2004). Sustainable development tends to be an "umbrella concept" for a holistic vision for a future welfare society and the emphasis is focused on the process of the framework involving economical development, social welfare, co-operation, good environment, democracy and participation, health and equity in a general sense (Lundqvist & Carlsson, 2004). Ellen Wohl (2001) put out the question if it is feasible to save a species without saving the ecosystem in which that species evolved, in the context of the historical degradation of rivers and associated restoration efforts. Törnblom & Angelstam (in press) put out the question if it is feasible to declare that a landscape or a catchment is sustainable and resilient as long as it meets societal needs and expectations, when society at the same time is loosing vital ecosystems and viable populations.

Riverine landscapes are dynamic at multiple spatial scales (Frisell *et al.*, 1986), and catchment scale processes are often considered as the most important drivers of system structure and function (Hauer *et al.*, 2003). There are thus suggestions to expand the assessments from isolated aquatic in-stream environment and the riparian zone, both of which are well studied (*e.g.*, Malanson, 1993; Bergquist, 1999), to the level of riverine landscapes constituted by entire drainage basins (Vannote *et al.*, 1980; Harding *et al.*, 1998; Wiens, 2002; Strayer *et al.*, 2003). To communicate this logic for planners and managers involved with policy implementation at different levels and in different sectors a two-dimensional approach is needed (Lazdinis & Angelstam, 2004). On the one hand, a multi-level approach to social and ecological systems including strategic assessment, tactical planning and operational management is needed (*e.g.*, see Lazdinis & Angelstam, 2004). On the other the implementing actors and institutions in a selected area need to be understood by 1) identification of the actors and mapping of policy networks, 2) evaluation of the implementation process to learn about the issues of concern, and 3) evaluation of policy implementation outcomes in the defined social-ecological system. This landscape approach must be carried out using both human and natural science approaches before applied at a variety of temporal and spatial scales from headwater stream segments to lowland river reaches, catchments and whole riverine landscapes. In Europe such an approach is explicitly advocated by the EU Water Framework Directive (Directive, 2000/60/EC), and implicitly by several other national and international policies

This thesis focuses on terrestrial and aquatic interactions within landscapes and catchments concerning macroinvertebrate distribution in catchments with different land use intensity and history, rapid assessments in streams using the Ephemeroptera-, Plecoptera-, and Trichoptera-group as a surrogate for species richness, thresholds concerning headwater catchment's proportions of forest and dose-response relationships concerning the macroinvertebrate Plecoptera group, predicted occurrence of viable freshwater pearl mussel populations, large woody debris correlations to brown trout and finally different governance approaches and a hierarchical tool box for achieving good ecological status in riverine landscapes. The read thread through this thesis is the effort towards understanding, communication and perception of how much is enough of habitats before loosing functional ecological integrity within landscapes and catchments at multiple spatial and temporal scales.

2 Objectives

The main objective of this thesis is to understand how much is enough of human impact before risking ecological integrity in different spatial scales, and propose a tentative framework for communication of possible thresholds that contributes to quantify the concept of "*good ecological status*" (Directive, 2000/60/EC). The hypothesis presented below spans from micro to macro scale structures and composition *i.e.* from in-stream habitat structures such as large woody debris, bottom substrates, and riparian composition to landscape- and catchment land cover composition. The specific hypotheses were:

1. Land use and cover explain the composition of biota in streams (**Paper I**)
2. Macro-scale variables contribute more than micro-scale riparian or in-stream variables to the explanation of macroinvertebrate assemblages in 2nd and 3rd order streams (**Paper I**)
3. Higher taxonomic levels of macroinvertebrates are surrogates for species richness in second and third order streams (**Paper II**)
4. The EPT-concept (Ephemeroptera, Plecoptera and Trichoptera) is a useful surrogate for species richness within headwater catchments (**Paper II**)
5. Plecoptera is an effective bio-indicator in headwater catchments for predicting the ecological integrity associated to terrestrial land cover (**Paper III**)
6. There are thresholds concerning forest proportions within catchments for Plecoptera abundance and taxa richness (**Paper III**)
7. It is possible to predict freshwater pearl mussel population viability from a combination of GIS-data describing land use in the riparian zone, water chemistry and host fish abundance (**Paper IV**)

8. The freshwater pearl mussel is an indicator species for landscape composition and catchment land cover in the riparian zone (**Paper IV)**
9. Large woody debris is correlated to the occurrence of brown trout (**Paper V**)
10. The abundance of large woody debris and brown trout are correlated (**Paper V**)

Finally, to reduce the observed scale mismatch between the need for systematic approach on the one hand and reality concerning the ambition to preserve all naturally occurring species in viable populations on the other, my ambition was to develop (**Paper VI**) an applied interdisciplinary, i.e. transdisciplinary approach (Tress *et al.*, 2005), thus implicitly supporting the implementation of policies about ecological integrity.

This thesis thus deals with how biota at different taxonomic levels respond to the composition and structure of ecosystems at multiple spatial scales from in-stream habitat structures such as large woody debris and bottom substrates to land cover composition in the regional landscape (Figure 1).

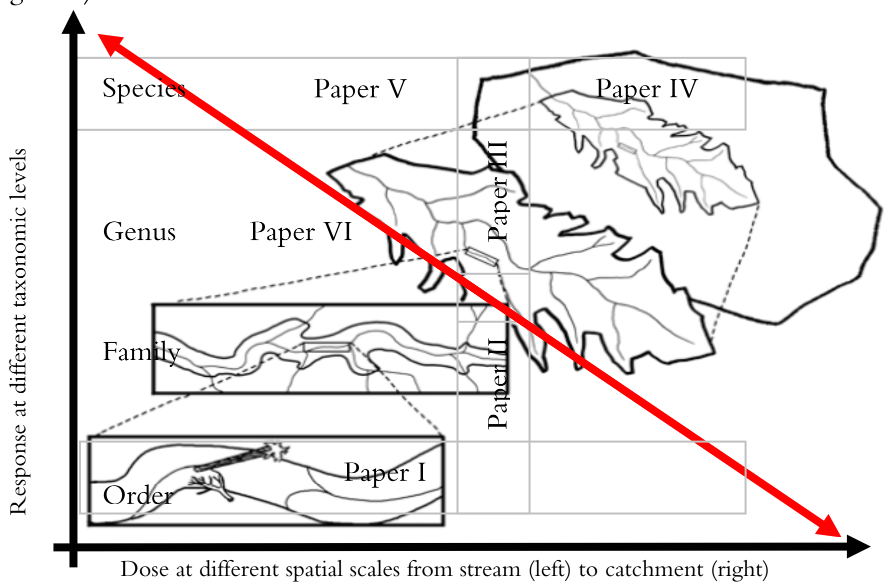

Figure 1. Illustration of how this thesis' six papers reflect the taxonomic level (vertical axis) and spatial scale (horizontal axis) associated with a landscape approach to support the implementation of policies about the maintenance of biodiversity and ecological integrity. Illustration Martin Holmer.

3 Background

3.1 Land cover composition and biodiversity in streams

Several ecological approaches have been developed to explain how streams function in the context of an entire catchment (Ward, 1989; Lorenz *et al.*, 1997; Harding *et al.*, 1998; Strayer *et al.*, 2003). These include approaches that focus on longitudinal and vertical changes of the biota with increasing stream order; that emphasize lateral interactions between terrestrial and aquatic dimensions; that integrate longitudinal, lateral and vertical dimensions of streams; or that stress spatial hierarchies and temporal change (Huet, 1954; Illies & Botosaneanu, 1963; Hynes, 1975; Vannote *et al.*, 1980; Newbold *et al.*, 1981; Stazner & Higler, 1986; Junk *et al.*, 1989; Poff *et al.*, 1997). Other concepts like landscape classification schemes (Frisell *et al.*, 1986) have been used for identifying and reducing uncertainties associated with natural variation in landscape processes. The use of such schemes was intended to minimize the number of parameters needed to categorize spatial variability and identify uncertainties (Bauer & Ralph, 1999). Stream classification systems use the same suite of key variables including geology, valley floor constraint, and channel slope to determine stream channel type over an entire catchment. Features used to identify common characteristics and classify channel reaches and valley segments are considered relevant for developing restoration plans, where stream reaches with similar physical characteristics may exhibit similar responses to land-use and restoration actions.

Variations in stream flow, invertebrate drift, boundary exchanges, patch context, or riverine connectivity affect different organisms in various ways (Wiens, 2002). Because different organisms have different movement capacities and expressions of patch or habitat selection, their responses to the heterogeneous structure of a landscape mosaic will differ. The overall patterns of composition, structure and function of biodiversity that occur within riverine systems reflect these organisms' responses to landscape structure in multiple scales. Species richness may be greater at ecotones or boundaries between patches in the riverine landscape (Ward & Wiens, 2001; Ward & Tockner, 2001), as a consequence of the attraction of some organisms to the boundary and the accumulation of others at the interface between hospitable and inhospitable patches (Wiens, 2002). Because different taxa may respond differently to landscape properties, the spatial patterns of diversity may also vary among groups. For example, Tockner, Schiemer and Ward (1998) documented a peak in fish diversity in portions of the Danube floodplain that had high connectivity to the main river channel, whereas amphibian diversity peaked where connectivity was low, in isolated floodplain ponds (Wiens, 2002).

These broad diversity patterns are ultimately founded on the ways in which particular organisms or species relate to landscape composition, structure and process. This leads to the general conclusion that riverine landscapes are proposed to be viewed from a multiple organismal rather than an exclusively anthropocentric perspective (Wiens *et al.*, 1993). However, there is always a risk in advocating an organism-based approach where there are tendencies for habitat-optimizing activities and situation-specific findings with little emergent generality. Describing broad patterns of biodiversity is one way to deal with this problem, but important information can be lost under the umbrella of "diversity" (Wiens, 2002). Several aquatic ecologists (*e.g.*, Townsend & Hildrew, 1994; Resh *et al.*, 1994; Rader, 1997; Poff *et al.*, 1997) have suggested that general patterns in the distribution and abundance of species or in the assembly of communities might be derived by aggregating taxa into "trait or functional groups" based on shared combinations of ecological and life-history features (Cummins, 1973; Cummins & Merrit, 1996). Similarly, the focal species concept was developed in response to the need to explore the opportunity of using particular species, guilds or functional groups as tools for biodiversity conservation (Roberge & Angelstam, 2004).

An important question pertains to understanding how natural variations (*e.g.*, disturbance regimes) in catchment and stream factors have determined watershed conditions (Benda *et al.*, 1998) in naturally dynamic ecosystems. The types, frequencies and extents of natural disturbances can be inferred using historical information about large-scale disturbance mechanisms (*e.g.*, flooding, fire and storm events) within relatively unaltered watersheds. Knowledge of natural disturbance regimes can facilitate assessments of how watersheds have been affected by human perturbations (Beechie & Bolton, 1999). However, some watershed processes have been altered by human actions for long periods of time over large areas, to the extent that natural disturbance regimes can no longer be identified (Bauer & Ralph, 1999). This stresses the need to cover the whole gradient of landscape alteration when studying ecological integrity within catchments.

In the first three papers of this thesis, I studied landscape composition, riparian vegetation, in-stream habitat characteristics and macroinvertebrate assemblages in 25 catchments located in the Carpathian Mountains in Central Europe. This region was selected because it presents much variation and a very long gradient in the intensity of land management activities known to affect ecological integrity of streams (*e.g.*, forestry and agriculture) among different areas belonging to the same ecoregion (Angelstam, 2006; Kuemmerle *et al.*, 2006).

3.2 Habitat variables and focal species' requirements

Milner *et al.* (1985) defined habitat for fish as the "local physicochemical and biological features of a site that constitute the daily environment for fish". Hence, although fish clearly respond to local conditions within the stream channel, habitat quality is also influenced by activities and conditions that may occur far from the stream.

With policies aiming at striking a balance between use of natural resources and biodiversity conservation within different catchments in a riverine landscape it could be essential that quantitative requirements for *e.g.* population viability are well understood by actors and stakeholders. A number of studies performed in various systems suggest that there may be thresholds in the biological response to habitat alteration (Carlson, 2000; Roberge & Angelstam, 2004; Angelstam *et al.*, 2005). Conditions well above ecological thresholds are sustainable or "healthy" (Haskell *et al.*, 1992), and conditions well below are unsustainable. Thresholds are rarely distinct. Rather they are intervals of change where, for example, a species or function changes from one state to another (Guénette & Villard, 2004). Theoretical studies of landscape patterns have identified critical thresholds in the abundance of particular habitat that produces qualitative differences in habitat connectivity (*e.g.*, Gardner *et al.*, 1987; Pearson *et al.*, 1996) or spatial processes that move across a landscape (*e.g.*, Turner, 1989). Empirical support exists for the presence of critical thresholds in habitat abundance on animal species in terrestrial landscapes (*e.g.*, Andrén, 1994; Angelstam *et al.*, 2004). Whether there are similar thresholds widely applicable to aquatic, semi-aquatic and riparian fauna remains unknown, possible threshold effects are not mentioned in any of the hierarchical riverine landscape concepts.

To derive performance targets for conservation and restoration, it is important to be aware of the environmental history of human actions and changes in ecosystems. Numerous environmental histories of watersheds are proving valuable because they demonstrate how natural and altered areas function and how they interact with riparian and fish habitats (Sedell & Everest, 1990; Lamberti *et al.*, 1991; Smith, 1993; Sear *et al.*, 1994; Wissmar & Beer, 1994; Reeves *et al.*, 1995). Historical perspectives can also help change society's perception about the degree to which today's stream and fish habitats are similar to historical ones. For example, younger people often tend to believe that what they see now is how it has always been (Wissmar, 1997). This may lead to accepting status quo of environmental conditions and issues as norms, both now and for tomorrow. Hence, a lack of information about historical conditions may perpetuate our unawareness of a continually changing environment (Harding *et al.*, 1998). Retrospective studies and the increased appreciation of past conditions that it brings, together with the analysis of remote reference conditions, are central to improving environmental communication and education (Wissmar, 1997).

3.3 Large woody debris as a functional in-stream structure in small forest streams

The production of fish and benthic fauna in forest streams is naturally based on the riparian supply of organic matter and nutrients from forest habitats in the catchment, and dimensioned by the riparian forest regulation of stream flow, temperature, insolation, and sediment load. Riparian forest also supplies forest streams with large woody debris (LWD), a key structure of stream ecosystems in temperate forested ecoregions (Kail & Hering, 2005).

Several studies have demonstrated the effects of LWD on the habitat and hydrodynamics of forest streams. In-stream structures as LWD can affect channel morphology by flow deflections creating scour pools, decrease distances between pools (Beechie & Sibley, 1997), and increase total pool area (Roni & Quinn, 2001) that occasionally affects sediment and debris deposition (Wallace *et al.*, 1995). This leads to increased nutrient retention in streams (Valett *et al.*, 2002), and further stabilization of stream banks and channels (Tschaplinski & Hartman, 1983), as well as increased habitat diversity (Naiman *et al.*, 1992). LWD influences stream hydrology, hydraulics, sediment budget, morphology and biota across a wide range of spatial and temporal scales (Harmon *et al.*, 1986; Gurnell *et al.*, 1995;

Gregory *et al.*, 2003). Large woody debris is also important for salmonid production, mainly due to increased habitat diversity (Fauch & Northcote, 1992; Flebbe & Dolloff, 1995; Degerman *et al.*, 2004). However, I do not know what the quantities of in-stream structures like LWD or dead wood are in naturally dynamic benchmark ecosystems, nor the extent to which different species are dependent on LWD in streams. A study in Swedish forest streams showed that levels of LWD in undisturbed sites generally was of the same order as in natural forests, and only 10% of Swedish forest streams had this amount of LWD (Degerman *et al.,* 2005).

The amount and quality of LWD in streams in Sweden has not been studied to the same extent as in terrestrial forest systems (Degerman *et al.*, 2004; Dahlström, 2005). Few studies on LWD exist for Scandinavian streams (Bergquist, 1999; Siitonen, 2001). However, there is evidence suggesting relationships between the age of the riparian forests and the amount and quality of LWD in the streams (Enetjärn & Birkö, 1998: Liljaniemi *et al.*, 2002; Lazdinis & Angelstam, 2004; Dahlström & Nilsson, 2006). It has even been suggested that LWD could be an in-stream structure that is limiting trout populations on a large scale in Sweden (Näslund, 1999). In Japan, Inoue and Nakano (1998) noted that the density of Masu salmon (*Onchorhychus masou*) was directly correlated with the amount of woody debris.

Degerman *et al.* (2004, Paper V) show that the occurrence and size of the largest trout were higher at sites with LWD present than at sites without LWD. This indicates that LWD creates a suitable environment for brown trout, probably by providing a station sheltered both from predators and water current (Tschaplinski & Hartman, 1983; Fausch & Northcote, 1992), and possible by creating pools, a habitat that generally holds larger trout than other habitat types (Heggenes, 1988; Näslund *et al.*, 1998).

3.4 Knowledge gaps for applying a hierarchical approach to assess ecological integrity

To date, efforts to restore and protect rivers in a general context have focused primarily on two goals – improving water quality (i.e., water chemistry), and establishing minimum flow requirements so that rivers and streams do not run completely dry (Postell & Richter, 2003). These actions have improved river conditions in many locations. Many fish populations are benefiting from less-polluted, less acidic waters. But the focus on minimum flows and water quality has done too little to restore the functions and processes that sustain the integrity of river systems overall (Postell & Richter, 2003).

The most direct and effective measure of the integrity of a water body is proposed to be the status of its living systems (Karr & Chu, 1999). These systems are the product of millennia of adaptations to climatic, geological, chemical, and biological factors. Their existence integrates everything that has happened where they live, as well as what has happened upstream and upland. During the last decade, scientists have gathered considerable evidence that a river's natural flow regime – its variable pattern of high and low flows throughout the year as well as across many years – exerts great influence on river health (Dynesius & Nilsson, 1994; Poff *et al.*, 1997). Each natural flow component performs valuable work for the system as a whole. Flood flows cue fish to spawn, provides new migration routes, and trigger certain insects to begin a new phase of their life cycle, for example, while very low flows may be critical to the recruitment of riverside (or riparian) vegetation.

Consequently, restoring rivers now under heavy human control requires much more than simply ensuring that water is in the channel or just putting back some stones or spawning gravel. Instead it is necessary to re-create to a sufficient degree the natural flow pattern that drives so many ecological processes and the natural dynamics of rapids and pools. Flow restoration may involve operating dams and reservoirs so as to mimic a river's pre-dam highs and lows. In rivers not yet heavily dammed or controlled, many of which are found in developing countries, the challenge is to preserve enough of the natural flow pattern to maintain ecological functions even while the river is managed for other economic purposes. Problems of past restoration can be attributed to planning and implementing projects that do not meet the fundamental characteristics and definition of restoration – a holistic process aimed at re-establishing ecosystem structure and function. Prior experience indicates a need for broader, catchment-scale restoration (Williams *et al.*, 1997; Rabeni & Sowa, 2002).

Several riverine landscape concepts have pointed out that land-use patterns affect both the form and the function of the rivers throughout the world, yet these effects are little recognized or understood. Still, no one can quantify how a particular land use, or an in-stream structure as LWD, impact stream biota. Apparently there are also severe gaps in understanding and monitoring of the spatial and temporal dynamics in different stream segments within a whole catchment. Yet, there exist some experiences from salmon restoration projects that reveal important insights of a more holistic catchment oriented perspective (Fausch *et al.*, 2002).

By using spatially explicit habitat models (*e.g.*, Scott *et al.*, 2002) based on habitat performance targets for a suite of forest dwelling focal bird species it has been possible to present assessment of habitat network functionality to forest managers and forest companies concerning the need for conservation and restoration management (*e.g.*, Angelstam *et al.*, 2003a). This has yet to be done for aquatic environments. However, this requires research on how much is enough of different structures and processes at multiple spatial scales to maintain viable populations of species.

4 Methods

4.1 Study areas

To carry out dose-response studies and thus contribute to quantifying the concept of "*good ecological status*" (Directive, 2000/60/EC) studies were made in Sweden, Poland, Ukraine and Romania at three spatial scales, viz. landscape, riparian zone and in-stream. I used a natural experiment design (*sensu* Diamond, 1986). Natural experiments differ from field experiments and laboratory experiments in that the experimenter does not establish the perturbation but instead selects sites where the perturbation is already running or has run. The perturbation may have been initiated naturally or by humans other than an experimental ecologist. Along with the experimental sites, the investigation selects control sites so that the two types of sites differ in presence and absence of the perturbation but are as similar as possible in other respects.

In **Papers I, II** and **III,** I use data from 25 individual 2^{nd} and 3^{rd} order streams in Central Europe's Carpathian Mountains. Using topographic maps I selected streams with similar catchment size, stream order and altitude, but with different degrees of naturalness and cultural authenticity. A total of 5 streams were studied in northern Romania and western Ukraine, respectively, and 15 streams in landscapes with different land management histories in southeastern Poland. Streams varied from low gradient, sand and silt, to high-gradient, cobble bottom. The size of these watersheds ranged from 0.5 to 7.8 km^2.

The freshwater pearl mussel dataset consists of 111 streams with freshwater pearl mussel populations in the County of Västernorrland, central Sweden (**Paper IV**) (method, is described below). Mussel surveys were performed during the period from 1990 to 2004 mainly during the summer months (June-August) by the local County Board. The main method used for sampling freshwater pearl mussel was on visual search in wadable streams for specimens using a water glass at suitable stream sites (*e.g.*, "*informal sampling*", see Strayer & Smith, 2003), with the ambition to cover the FPM populations distribution within a stream system (SEPA, 2007). The main purpose of this survey method is to monitor eventual change of population size, abundance, and changes in age-/size structure in separate populations of the freshwater pearl mussel (SEPA, 2007).

Data on large woody debris (LWD) and fish in streams were compiled from the Swedish Electrofishing RegiSter (SERS), which is a database with over 10,000 studied sites (**Paper V**). The amount of LWD was quantified at 4,382 forest stream sites. Only sites with riparian forest classified as coniferous, deciduous or mixed forest were included. The sites were located at altitudes of 1-895 m a.s.l. (average 175 m.a.s.l), and distributed all over Sweden.

To operationalise the idea of a landscape approach encompasses entire landscapes as social-ecological system a transdisciplinary hierarchical toolbox for achieving "*good ecological status*" in riverine landscapes is discussed and problemized in the context of the increasing amount of EU legislations that each member state has to comply with to achieve ecological integrity within catchments (**Paper VI**).

4.2 Land cover composition and macroinvertebrate assemblages (Paper I)

In each of the 25 streams, ten riffle sites (0.04 m^2) over a section at least 50 m long at the lower end of the catchment were examined. To sample invertebrates the substrate was disturbed by a hand brush within the frame of a standard Surber sampler. The specimens were determined to taxonomic order, as it has been suggested that a higher-level taxonomic and ecological structure usually provides a better guideline for classification than primarily focusing on species (Karr & Chu, 1999).

The land cover types in the catchments were determined using field surveys in accordance with the Swedish grassland inventory program and European Common Agricultural Programme (Naturvårdsverket *et al.*, 1996; Ihse & Lindahl, 2000; Statens Jordbruksverk, 2005). These approaches were chosen to assess the cultural authenticity of landscapes dominated by wooded grasslands. The surveys were made in transects 500 m wide and 2000 m long from the bottom of the valleys and up on the valley sides. These surveys covered most of the catchment areas and were used as the basis for the land cover type classification of Landsat imagery (LandsatTM, Path 186, Row 26, 1998-07-31and LandsatTM, Path 185, Row 27, 1998-08-09). The catchments were delineated according to topographic maps with a scale of 1:100 000, and the delineations were digitised with ArcGis 8. The total areas as well as the areas of the land cover types within the catchments were then calculated.

The riparian zone was defined as a corridor on both sides adjacent to the stream that includes all bankside and closely surrounding vegetation (Giller and Malmqvist 1998). Two separate widths were used of riparian zone (5 and 30 m) on either side of the stream over a 50 m range along the stream based on recommendations by the Swedish Environmental Protection Agency (SEPA, 2003). The dominant and sub-dominant land types or land use types were classified. If the characteristics of the zones on each side of the stream were very different, or if they were present to the same extent, both types were documented (SEPA, 2003). Also, a narrower riparian zone (0-5 m) was defined as the closest 5-m wide riparian strips on both sides of the stream over a 50 m range along the stream. Riparian vegetation consists of trees, shrubs, tall herbs and other vegetation. If the riparian zone lacked vegetation or included more than 50 % cultivated land, it was classified as "others". Dominating and sub-dominating riparian land cover types and structures were also measured.

In-stream conditions, including the aquatic biotope, were described according to a modified version of SEPA (2003). Measurements were made on water current, bottom substrate (inorganic and organic material) turbidity, shadow, site length, site area, average and maximal depth of the site, stream order, catchment's area, altitude and large woody debris (LWD) and fine woody debris (FWD).

From every stream one grab sample of water, from the midsection of the stream, was collected in a 250-ml plastic bottle prior to collecting biota samples. Conductivity was measured in the field along with temperature. In the laboratory UV/Vis absorbance spectra in the range 190-1100 nm were recorded with a spectrophotometer using a 1 cm quartz cuvette on unfiltered samples and at ambient pH. Absorbance values at 254 nm (ABS254) and 420 nm (ABS420) were used for further data analysis. Alkalinity was measured according to end-point titration with HCl (0.02 M) to pH 5.6 (ISO 1995). Total carbon (TC) and inorganic carbon (IC) concentrations were quantified, and total organic carbon (TOC) was calculated as the difference between TC and IC.

To directly associate taxonomic composition with multivariate environmental gradients, an ordination was performed with Canonical Correspondence Analysis (CCA; ter Braak, 1989). The CCA assumes that the abundance of species is a symmetrical unimodal function of position along environmental gradients. The selection of environmental variables to be included in the ordination was done using CCA analysis with Monte Carlo simulation. This identifies the variables which contribute most to the variability of the species data and which should therefore be retained in the final CCA. These variables were also used in Principal Components Analysis (PCA) to evaluate correlation with other environmental variables. In the final CCA Monte Carlo randomization was used to break the relationship between the environmental data and the species. The order of samples (sites) in the species array was randomised. By repeating this procedure 10,000 times, calculating the probability that the observed magnitude of the Eigenvalues of each CCA axis was produced by chance. Ecom Software version 3.0 from Pisces was used for CCA, while SPSS (SPSS inc.) version 12 was used for the remaining statistical tests.

4.3 Rapid assessment of macroinvertebrate species richness in second and third order streams (Paper II)

In this study three alternative methods for rapid assessment (Gordon *et al.*, 2004) of macroinvertebrate species richness in running waters were evaluated, focusing on the Ephemeroptera, Plecoptera and Trichoptera orders, which is a group of orders widely used in biomonitoring (Lenat, 1988; Barbour *et al.*, 1999; Sandin & Johnson, 2000). Macroinvertebrate specimen were determined to family, genus and species level, and collected across a gradient of land-use intensity in the Carpathian Mountains to evaluate three shortcuts for surrogates in the assessment of macroinvertebrate species richness in second and third order streams: (1) the use of data in higher taxonomic levels, (2) the use of species-level data from EPT-indicator orders, and (3) the use of abundance data. Finally the relationship between land cover proportion and taxa richness for Ephemeroptera, Plecoptera and Trichoptera was evaluated in order to cast light on the possible drivers of diversity in those orders.

4.4 Plecoptera as a bioindicator for catchment integrity (Paper III)

This study tested if Plecoptera taxa richness and abundance was related to the forest cover of catchments. Reference conditions for the catchment's proportion of forest was established by choosing five catchments located in nature reserves or National Parks representing unaffected or near pristine conditions with high proportions of forest (catchment's forest proportion > 90%). By descriptive statistics estimations of the lower bound within a 95%

confidence interval for mean of Plecoptera taxa and for Plecoptera abundance of individuals was possible. This interval was used as a cut off value, distinguishing areas of higher and lowers ecological integrity. Using binary logistic regression it was possible to test if forest proportion in the catchment could be indicated by Plecoptera taxa or abundance. By using the variables in the equation from the logistic regressions it was possible to formulate thresholds for expected outcomes for catchment's forest proportion associated to the evaluated dependent variable and conductivity.

4.5 Predicting occurrence of viable populations of freshwater pearl mussels, *Margaritifera margaritifera* (L.) (Paper IV)

The hypothesis that it is possible to predict freshwater pearl mussel (FPM) population viability in 111 Swedish streams was tested. By using a combination of data describing land cover and use within the riparian zone covering 50 meters on each side of the streams from their local distribution up to the headwater sections, water chemistry and electrofishing data describing the abundance of host fish species for mussel larvae it was possible to identify a number of key variables for predicting viable populations of the freshwater pearl mussel. The freshwater pearl mussel was also tested for the potential candidate as an indicator species for subtle impacts as land use.

Mussel surveys were performed by the County Administration Board of Västernorrland mainly during the summer months. The main method used for sampling FPM is described in detail in SEPA (2007), and is based on visual search in wadable streams for specimens using a water glass at suitable stream sites (*e.g.,* "*informal sampling*", see Strayer & Smith, 2003), with the ambition to cover the FPM populations distribution within a stream system. The main purpose of this survey method is to monitor eventual change of population size, abundance, and changes in age-/size structure in separate populations of the freshwater pearl mussel (SEPA, 2007). In the prescribed method the FPM status was ranked according to 6 classes of population structure. This classification, called Mussel status V, has been developed by the County Board of Västernorrland, based on empirical evidence and on earlier studies (Young *et al.*, 2001). For the statistical analysis also a two-graded status scale (Mussel status II) was constructed by merging classes 1 and 2 (classified as viable populations) into one class, while classes 3, 4 and 5 (classified as not viable – soon extinct) were merged into a second-class.

Approximately half of the sites (n=58 of 111; 52%) were electrofished by wading using dead current equipment according to Swedish standards (Degerman & Sers, 1999). The investigations were carried out by the

County Board or by the local office of the Swedish Fishery Board in Härnösand. When data from several electrofishing occasions were available, data that closest matched mussel inventory in time and distance (maximum of 5 km) was selected. Being the host species for mussel larvae, the focus was on data concerning brown trout, and the occurrence of yearlings (0+) and older brown trout (> 0+) per 100 m^2. A calculation of fish abundance was made according to Bohlin *et al.* (1989) for multiple runs and according to Degerman & Sers (1999) when only a single run was carried out.

Water samples for chemical analysis were collected in winter (base flow) and spring (snow melt) as part of the county board's freshwater monitoring program. The variables pH, alkalinity (meq/l), colour (mg Pt/l), conductivity (mS/m), Ca+Mg (meq/l), P_{Tot} (ug/l) and turbidity (Fnu) were measured using standard methods.

Catchment variables such as stream slope at the pearl mussel site (m/km), distance to nearest upstream lake, and estimated area of first nearest upstream lake (> 5 ha) were derived using ArcView 9.2 GIS (Geographical Information System). Catchment land cover was measured in 50-m wide buffers (the riparian zone) upstream the freshwater pearl mussel sites' covering all the upstream reaches in the headwaters. Land use was classified as coniferous, deciduous, and/or young forest, clear-cut, agricultural land, bogs and pastures according to Swedish CORINE Land Cover (SCLC) version 2.3 (European Commission, 1993). All land use variables were expressed as the proportion of the whole riparian buffer zone. Catchment land use classifications were made using GIS-data from 2000, whereas mussel surveys were from 1990-2004, however most land use activities have not been changed.

4.6 Dead wood in Swedish small forest streams (Paper V)

Large Woody Debris (LWD) was defined as having a diameter of 10 cm or more and a length of at least 50 cm. The number of pieces of LWD was counted in the site and presented as pieces of LWD 100 m^{-2}. The sampling sites were selected in areas with a habitat suitable for spawning and the first years of growth of brown trout. Electrofishing was carried out in August-September by wading, using dead or pulsed dead electric current. The average length of stream sampled was 46.8 m and the average width was 6.8 m. The average sampled stream area was 238 m^2. Fish were determined to species, and total length of fish individuals was measured. Population densities were estimated according to Bohlin *et al.* (1989) if consecutive runs had been carried out. Otherwise densities were estimated from average catch efficiencies for the species and age group (Degerman & Sers, 1999).

The environmental stream variables registered were width, mean depth, maximum depth, and dominating and sub-dominating substrate. The substrate was classified in five categories based on the dominating particle size. Water velocity was classified into three classes at sampling in late summer flow situations. The size of catchments and the proportion of lakes within the catchments were measured on topographic maps.

4.7 Towards a hierarchical toolbox for TerrAquatic perspectives in riverine landscapes (Paper VI)

Sustainable development and biodiversity are two contemporary concepts closely linked to the use and management of natural resources. There is, however, an urgent need to operationalise ecological sustainability and to include this into governance and planning processes at multiple levels and across sectors. Provided that policies are explicit this can be done by (1) translating policy contents to measurable variables and by applying performance targets that define ecological sustainability on the one hand, and (2) to develop local and regional governance arrangements for the maintenance of ecological sustainability in terms of the composition, structure and function of ecosystems on the other (*e.g.*, Törnblom & Angelstam, in press). This requires syntheses of existing experiences and research on how different elements of sustainability can be defined, assessed, integrated and communicated among actors and stakeholders. Important dimensions to consider when designing such accounting systems are (1) the systems of governance and planning, (2) the historical development of landscapes (i.e. social and ecological systems), as well as (3) the type of actors and stakeholders.

It is well established that river ecosystems function at multiple spatial scales (Frisell *et al.*, 1986). Contrary to common belief, landscape scale and not riparian and instream processes is the most important driver of ecosystem structure and function (*e.g.*, Hauer *et al.*, 2003). However, although this understanding is well established in the scientific community, managers seldom incorporate this knowledge in their attempts to conserve or restore lotic systems (*e.g.*, Sabatier *et al.*, 2005a). Indeed, the history of river restoration, as other kinds of ecological restoration, is largely marked by ineffectual attempts directed at the wrong spatial scale (Kershner, 1997). Often, restoration efforts focus on site-specific projects within small areas, or with a single disciplinary focus narrowly defined or confined to a specific species that may be threatened or endangered, or to a species with a sport or commercial interest (Hauer *et al.*, 2003), thus disregarding the important linkages and connectivity between the river and its surrounding landscape.

Accordingly, there is now a recognized need to expand the assessments from the instream environment and the riparian zone, both of which are well studied (*e.g.*, Malanson, 1993; Bergquist, 1999), to the level of riverine landscapes constituted by entire catchments (Wiens, 2002; Sabatier *et al.*, 2005a, b).

Moreover, because landscapes are integrated social-ecological systems (*e.g.*, Berkes *et al.*, 2003) there is a strong need to develop a multi-level toolbox for the social systems' planners, managers and stakeholders that includes all steps in the adaptive governance and management cycle towards policy implementation, including governance, management, and assessment (Meffe *et al.*, 2002; Lazdinis & Angelstam, 2004).

Specifically, a comprehensive tool-box is needed, which can be applied to both ecological and social system dimensions of catchments (Figure 2). This includes systematic analyses of (1) of ecosystems at a variety of temporal and spatial scales from headwater stream segments to lowland river reaches, and from catchments to riverine landscapes and entire drainage basins, and (2) of social systems' actors and stakeholders representing multiple levels of governance and different sectors affecting the state and trends of ecological integrity in catchments.

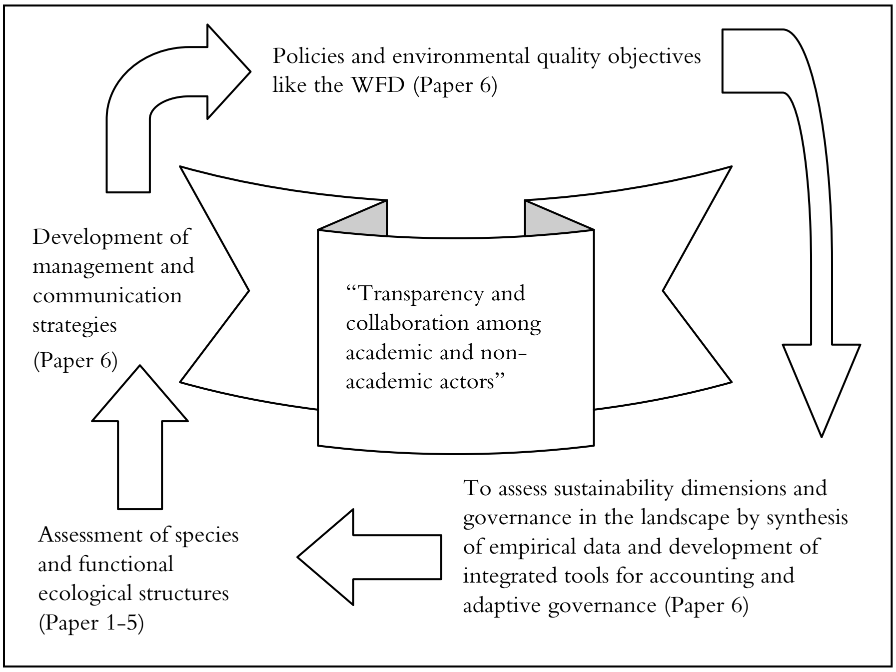

Figure 2. Illustration showing a simplified policy cycle (see Mayers & Bass, 2004) and the multi-level toolbox presented in paper 6, which concerns both ecological and social dimensions of catchments at multiple scales**.** Paper 6 presents a hierarchical iterated approach within an adaptive governance and management framework for involving research, synthesis, and public participation in adaptive governance and ecosystem management from headwaters to the sea from sub-catchments to landscapes within drainage basins indicating different levels of ecological integrity, history and socio-cultural perspectives.

5 Results and Discussion

5.1 Paper I

I found clear relationships between four land cover types and the composition of macroinvertebrates. These results suggest a stronger relationship between landscape level characteristics and the in-stream fauna composition, than local riparian or in-stream conditions and the fauna composition. The low taxonomic resolution (order) could be responsible for the fact that only large-scale landscape variables were significantly affecting the taxa composition. With higher taxonomic resolution, the taxa composition would more probably reflect key environmental or biological limiting factors for single species.

Canonical correspondence analysis suggested that variation in taxa assemblage structure was primarily related to four land cover types at the landscape scale. These were the proportions in the catchments of (1) broadleaved forest, (2) fine grained agricultural areas with trees (meaning pre-industrial cultural landscape), (3) mixed forest and (4) natural grassland without trees, respectively (Figure 3). Principal Component Analysis (PCA) indicated that landscape composition and in-stream bottom substrate co-varied. There were finer sediments in streams in the open agricultural landscape, and the distribution of fine sediments as a bottom substrate were suggested to depend on the types of anthropogenic activities associated to the open agricultural landscape. The PCA also showed that all studied chemical variables, including organic carbon, had higher values in the agricultural landscape relative to natural forests.

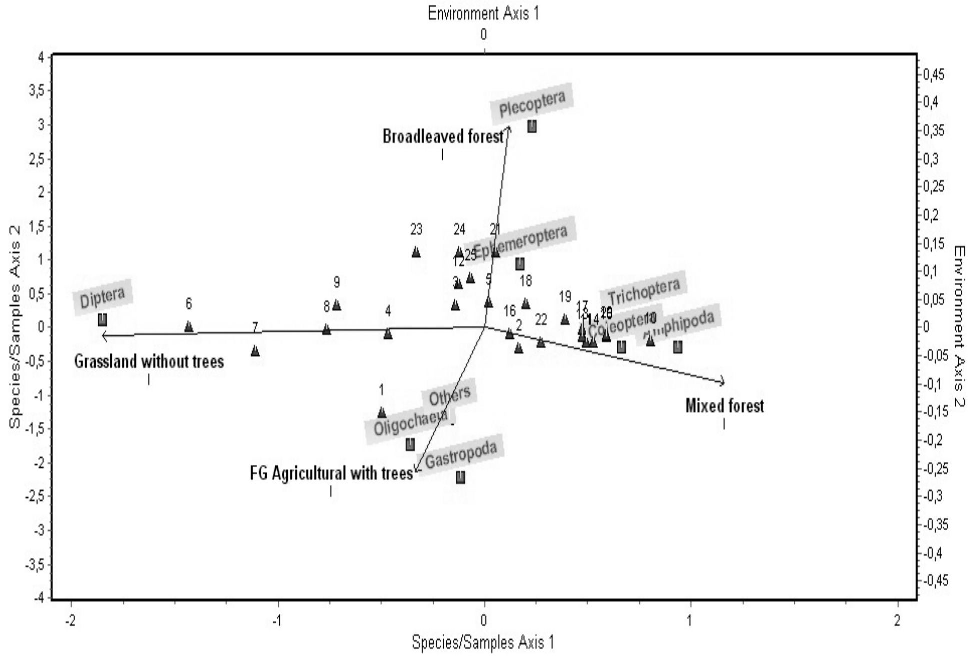

Figure 3. Canonical Correspondence Analysis ordination of nine taxonomic groups on four catchment variables. Plot of the first two significant axises. Vectors (catchment variables) shown as arrows, taxonomic groups or class as squares and sites (1-25) as triangles.

The major source of variation among taxa in the streams was a higher abundance of *Diptera* in agricultural landscapes against *Plecoptera*, *Coleoptera*, *Trichoptera* and *Amphipoda* in forests. *Gastropoda* and *Oligochaeta* were more abundant in open fine-grained agricultural landscapes with trees. *Ephemeroptera* were quite indifferent to these gradients in catchment land cover, but showed a tendency for being more abundant in open landscapes with trees.

The results suggest that the regional natural history and land cover could provide a guide for site classification and for understanding connection between human-induced changes in land use and cover on the one hand, and the distribution of macroinvertebrate assemblages at the level of taxonomic orders on the other. In most landscapes, or catchments, geomorphology determines the soil and availability of ions, and the slope, soil and climate determines the vegetation (Moore *et al.*, 1991; Chipman & Johnson, 2002), which also determines the supply of organic matter together with the soil that influences water chemistry and water inputs to the stream

(Minshall *et al.*, 1985; Naiman *et al.*, 1987). Human activity in the catchment affects streams too, leading to large effects on streams and rivers, sometimes direct and obvious, sometimes more subtle (Giller & Malmqvist, 1998). My results do not contradict this statement. Indeed four land cover variables explained 54.8% of the variation in macroinvertebrate community structure. This leads to the conclusion that I cannot exclude that human land use within the Carpathian catchments is reflected in the in-stream macroinvertebrate distribution. However, it is very possible that a higher taxonomic resolution would provide more detailed information about ecological mechanisms and functionality when species and functional groups are considered as compared with only taxonomic groups. Thus, in order to understand human activities and their more subtle effects on stream biota there are knowledge gaps concerning the effects of composition and structure of land cover types, riparian habitats and in-stream conditions.

5.2 Paper II

Using data from 25 headwater streams in the Carpathian Mountains in Central Europe revealed that the use of the EPT-metric, as an indicator of ecological integrity and diversity, at genus and family levels within macroinvertebrate orders in headwater streams was promising, except for Ephemeroptera. By contrast, species-level data and abundance of individuals from indicator taxa across orders demonstrated that the EPT-group was not promising as indicator for macroinvertebrate richness in general. However, Trichoptera alone was a very good indicator of diversity within the EPT-group, but also to some extent for all macroinvertebrates. This study demonstrated that correlations for abundance of individuals and species are weaker than correlations between taxa at the genus and family level. There was no significant evidence for usefulness of abundance of EPT-species as a surrogate for species richness, except for Plecoptera and abundance for EPT species richness, and Ephemeroptera for species richness of all macroinvertebrates (Table 1).

Table 1. Relationships between species richness within Ephemeroptera, Plecoptera, and Trichoptera on the one hand, and genus richness, family richness and individual abundance within the respective orders on the other (n = 25).

Order	Mean (SD)	r_s species richness [1]
EPT[2]		
Species	18.0 (6.5)	-
Genera	17.2 (5.0)	0.947***
Families	12.6 (3.7)	0.941***
Abundance	323.2 (201.7)	0.595**
Ephemeroptera		
Species	6.4 (2.0)	-
Genera	5.0 (1.7)	0.922***
Families	3.2 (1.2)	0.655***
Abundance	192.7 (136.7)	0.315
Plecoptera		
Species	5.0 (2.9)	-
Genera	4.5 (2.5)	0.981***
Families	3.3 (1.7)	0.896***
Abundance	63.6 (82.9)	0.867***
Trichoptera		
Species	6.6 (3.1)	-
Genera	7.8 (2.4)	0.860***
Families	6.1 (2.2)	0.925***
Abundance	67.0 (87.6)	0.256

P-values: ** $P < 0.01$, *** $P < 0.001$.

[1] Spearman correlation coefficient for relationship with species richness within the evaluated order (or group thereof).

[2] Ephemeroptera, Plecoptera, and Trichoptera taken together.

Catchment land cover types as forest proportion was correlated to EPT species richness and family richness, but strongly correlated to Plecoptera taxa richness at species, genus, and family levels, and also to abundance of Plecoptera individuals. The cover of grassland and agriculture proportion was negatively correlated to EPT taxa richness at all taxonomical levels (Table 2). These results raise further questions for conservation of biodiversity issues in the context of reference conditions' concerning how much is enough of anthropogenic disturbance and human development before passing critical thresholds for ecosystem functionality and how the results should be communicated to the different actors of the society in order to make a difference.

Table 2. Spearman rank correlation coefficients for the relationships between taxon richness and individual abundance within Ephemeroptera, Plecoptera, and Trichoptera, and the proportions of three main land cover types in Carpathian catchments (n = 25).

	Spearman's correlation coefficient		
Order	**Forest**	**Grassland**	**Agriculture**
EPT[3]			
Species	0.499*	- 0.162	- 0.550**
Genera	0.380	- 0.090	- 0.499*
Families	0.412*	- 0.068	- 0.585**
Abundance	0.353	- 0.076	- 0.427*
Ephemeroptera			
Species	0.016	0.332	- 0.420*
Genera	0.036	0.206	- 0.355
Families	- 0.425*	0.466*	0.087
Abundance	0.041	0.206	- 0.319
Plecoptera			
Species	0.686***	- 0.537**	- 0.387
Genera	0.615**	- 0.456*	- 0.382
Families	0.614**	- 0.283	- 0.627**
Abundance	0.619**	- 0.274	- 0.609**
Trichoptera			
Species	0.281	- 0.029	- 0.368
Genera	0.142	0.091	- 0.314
Families	0.470*	- 0.173	- 0.552**
Abundance	0.383	- 0.554**	0.035

P-values: $^{*}P < 0.05$ $^{**}P < 0.01$, $^{***}P < 0.001$.

[3] Ephemeroptera, Plecoptera, and Trichoptera taken together.

5.3 Paper III

I found evidence for using Plecoptera as an effective bioindicator in headwater catchments for predicting the ecological status associated to terrestrial land cover. Plecoptera abundance and Plecoptera taxa richness were well correlated to each other as well as to catchments' forest proportion in 25 Carpathian headwater streams (Figure 4). Plecoptera taxa richness and abundance of individuals were negatively correlated to catchment area, inorganic carbon, alkalinity and conductivity. I evaluated Plecoptera abundance of individuals considering that counting Plecoptera individuals is easier for non-experts than recognizing different Plecoptera taxa, also abundance gave higher proportion of correct classification.

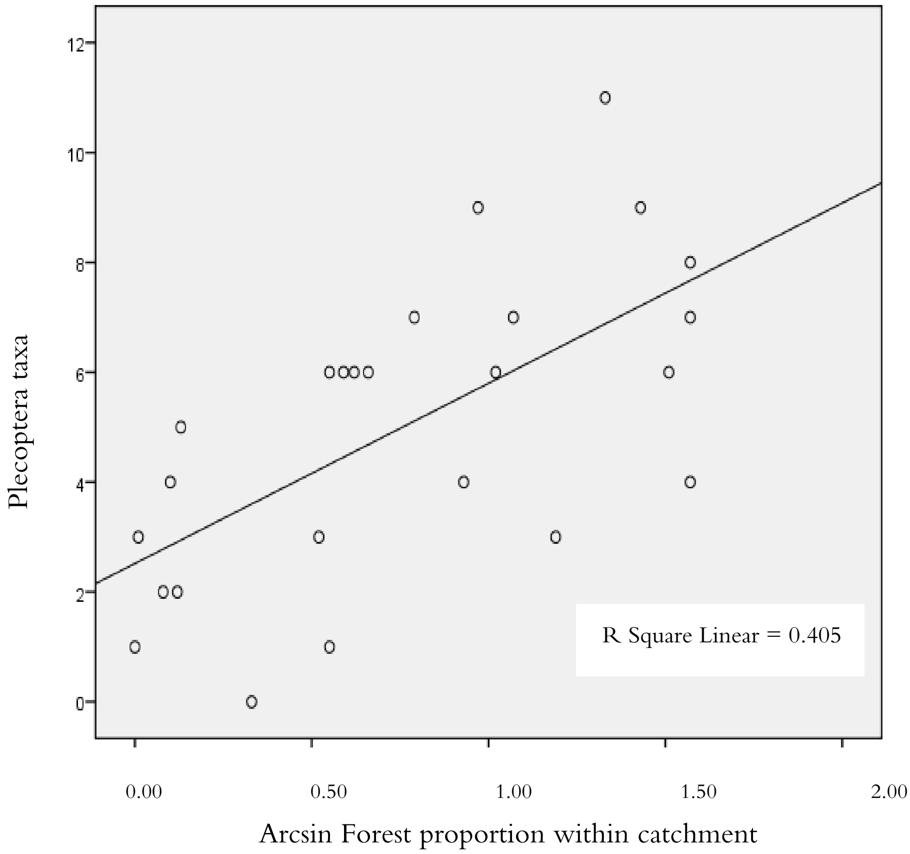

Figure 4. Pearson correlation of the relationship between Plecoptera taxa richness and Arcsin Forest proportion within 25 Carpathian catchments.

The results suggested that Plecoptera was not only correlated to forest proportion, also chemical constituents seemed important. At a fixed conductivity of 15 mS/m a forest proportion of 79% was the threshold value, separating catchments with a 0.5 probability of finding ≥ 64 Plecoptera individuals per ten Surber samples from catchments with likely fewer Plecoptera (Figure 5).

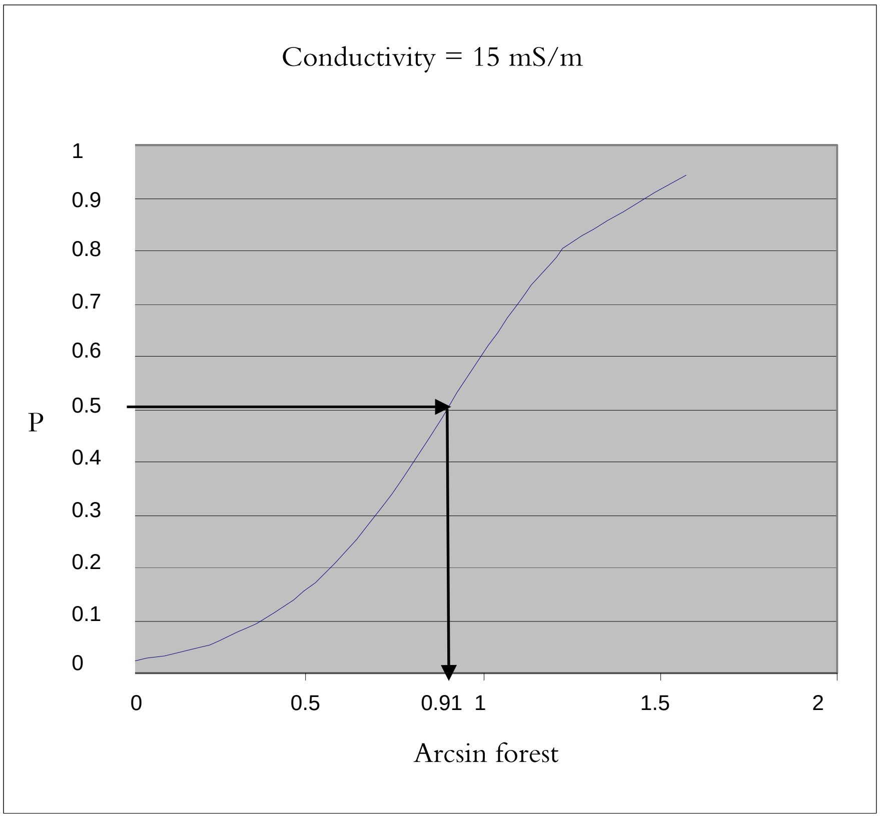

Figure 5. At Arcsin forest = 0.91 (*e.g.* forest proportion = 0.79, and Arcsin forest 1 ≈ 0.85% forest proportion), the probability = 0.51. A minimum of 79% forest proportion leads to the probability that the number of individuals of Plecoptera is 64 individuals per ten Surber samples. A multiple logistic regression using catchment forest proportion and associated conductivity (Z = -1.325 + (4.211 x Arcsin forest proportion) - (0.164 x conductivity)) to predict which streams held high abundance (≥ 64 individuals) of Plecoptera.

I argue that an indicator system well adapted to the EU Water Framework Directive's local water management plans and the general public should build on a suite of species with well-documented indicator and umbrellas value for each stream order type of conservation interest, which also have a high communication value for improving local participation for restoring ecological integrity in impaired headwater streams.

5.4 Paper IV

The freshwater pearl mussel *Margaritifera margaritifera* (L.) has become scarce and threatened throughout its distribution, and is consequently listed as "vulnerable" by IUCN. The hypothesis was tested that it is possible to predict freshwater pearl mussel population viability (status) from a combination of GIS-data describing land use in the riparian zone, water chemistry and electrofishing data describing the host fish species (brown trout; Salmo trutta) for mussel larvae. The data set covers 111 Swedish boreal catchments.

A good separation between populations could be achieved using only four variables; water colour (during spring flood < 80 mg Pt/l), turbidity (< 1 FNU), phosphorous (< 15 μg/l**),** and brown trout abundance (> 5 per 100 m^2) as criteria. If all chemical variables were below indicated threshold values and brown trout abundance above the threshold, 79% of mussel populations could be correctly classified (without the aid of statistical procedures).

Turbidity and phosphorous levels were correlated and associated to land use (Figure 6). The suggested mechanism is increased anthropogenic impact within the catchments leading to increased amount of nutrients, organic matter and sediment bed load in the stream substratum (Figure 7). It is suggested that this negatively impacts young mussels, probably by restricting oxygen levels and eventually by decreasing food availability and quality. Streams with upstream lakes held significantly lower levels of turbidity and water colour, lakes thus ameliorated the negative impact of land use in the riparian zone on mussels.

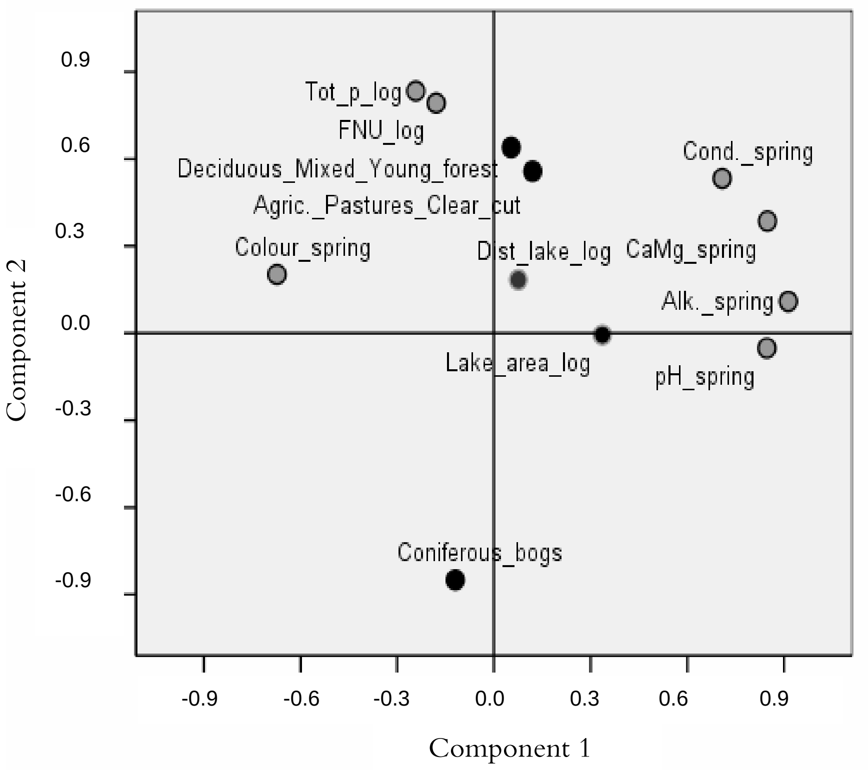

Figure 6. First two axes (components) of PCA on water chemistry, associated lake area and distance to upstream lake and surrounding land cover variables. Axis 1(Component 1) separates between acid/brown coloured streams and alkaline/clear streams (pH/COLOUR). Axis 2 (Component 2) separates between coniferous forest and affected ecosystems (IMPACT).

Using discriminant analysis on yearling brown trout density and land use in the riparian zone 80.4% of populations could be correctly classified as viable or without reproduction (absence of young mussels). The freshwater pearl mussel is evidently an indicator species for landscape composition and catchment land cover in the riparian zone.

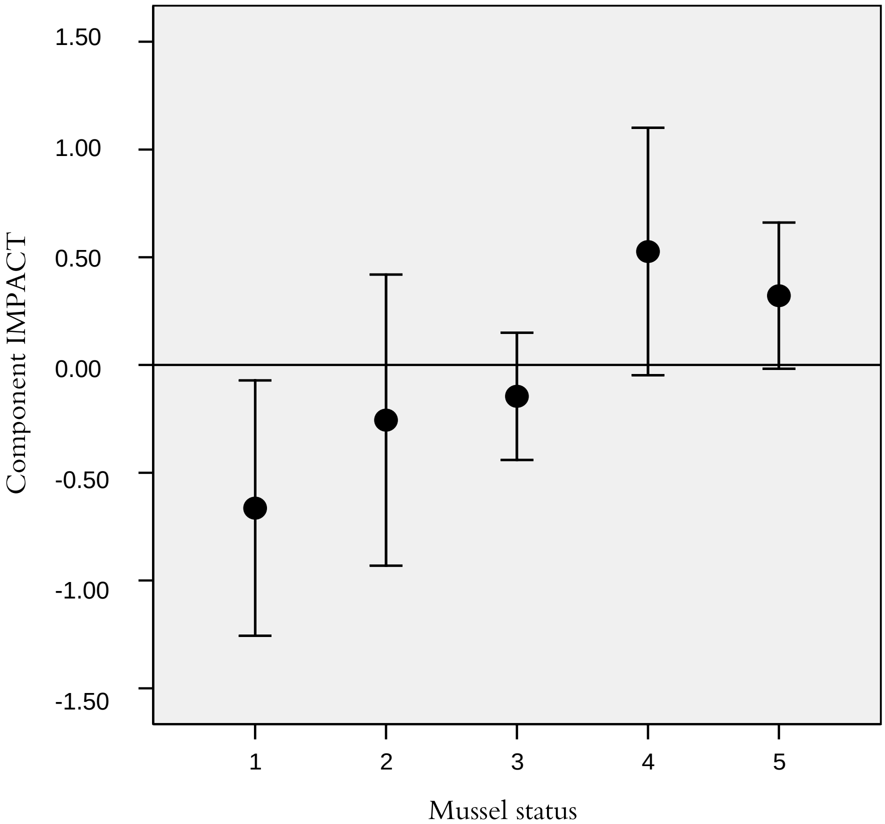

Figure 7. Positions along PCA-Component IMPACT (axis 1) distributed on five classes of FPM-status. High mussel status (1) was represented in streams with low impact in the riparian zone (large proportion of coniferous forest and bogs), while low status (5) was represented in streams surrounded by agricultural land, pastures and clear-cuts. Mean and 95 %-confidence interval is shown as dots and box plot interval (n=111).

With the FPM viability as an indicator, complemented with electrofishing on trout, water chemistry and GIS-data it is possible to quantify how large-scale land use impact the FPM, and further the ecological integrity of a stream system. This result also implies that there are needs to start manage and restore FPM-habitats in Swedish streams, not just only by protection and conservation of habitats, segments or reaches of streams. The waters are the mirror of the watershed (Hynes, 1975), and the complex life history of the freshwater pearl mussel makes it sensitive to land use and thus an excellent indicator of the ecological integrity and sustainability of waters affected by human activities. It is suggested that the thresholds indicated by FPM in this study also applies to other organisms, irrespectively of the presence of freshwater pearl mussels.

5.5 Paper V

Brown trout was the most common fish species in the investigated forest streams in Sweden, occurring in 82% of the sites. The occurrence increased with stream width and the brown trout abundance was highest in the smallest streams. Large Woody Debris (LWD) was present at 73% of sites, and brown trout occurred more frequently at sites with than at sites without LWD. The abundance of trout increased with LWD, and the effect was especially pronounced in sites with more than 4 pieces of LWD 100 m^{-2}. The abundance of trout increased with increasing amount of LWD up to 8-16 pieces 100m^{-2} (Figure 8), and by using the quantity of LWD and stream width, brown trout abundance could be predicted. Maximum size of brown trout caught at each site was correlated with LWD.

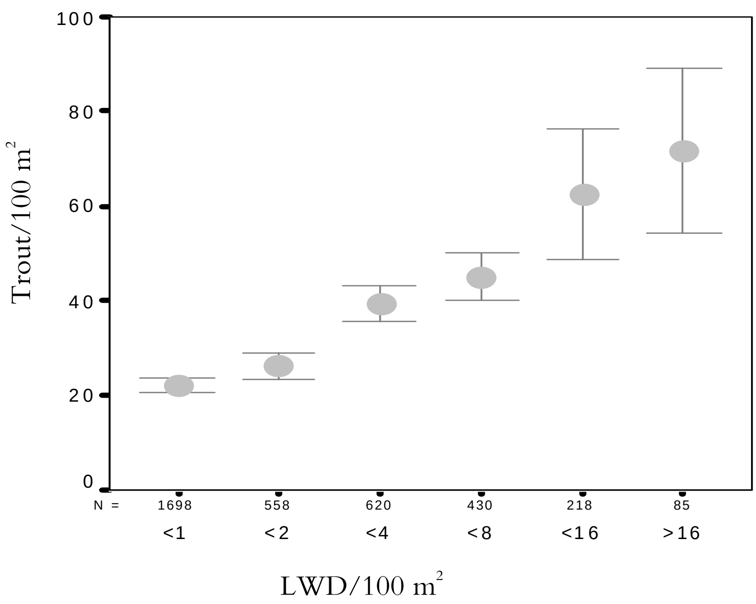

Figure 8. Abundance (no. 100 m^{-2}) of brown trout versus quantity of LWD. Bars indicate 95 % confidence intervals.

The largest trout caught averaged 188 mm at sites without LWD and 200 mm at sites with LWD, and this difference was significant. The occurrence and size of the largest trout were higher at sites with LWD present than at sites without LWD. This indicates that LWD creates suitable environments for brown trout by providing a micro habitat sheltered from water current and predators (Tschaplinski & Hartman, 1983; Fauch & Northcote, 1992; Näslund *et al.*, 1998), and possibly by creating pools, a habitat that has larger brown trout than other habitat types (Heggenes, 1988). Increased size, occurrence and abundance of trout with the amount of LWD indicate that suitable sites for foraging and refuges may be limiting factors for brown trout (Bachman, 1984).

In Sweden, for several decades, over 95% of the forested area is managed and subject to clear-felling systems without sound catchment management principles (Eckerberg, 1988). There is a lack of holistic and multidisciplinary perspectives in management of catchments that have been drained and are dominated by conifer re-forestation. There are also obvious gaps in the functionality of managed landscapes where processes like fire and flooding no longer maintain old forest and dead wood (Lazdinis and Angelstam, 2004). Despite the fact that several studies have described forestry's impact on stream ecosystems (Ramberg, 1976; Lynch *et al.*, 1977; Graynoth, 1979; Eckerberg, 1981, Borman & Likens, 1985; Gregory *et al.*, 1987; Chamberlin *et al.*, 1991; Lemley & Hildebrand, 2000), and that protection of riparian zones is of essential importance to fish in rivers and streams, the information has rarely been implemented in practice (Eckerberg, 1988). As a consequence, riparian forests have been harvested and the amount of LWD in the streams has been impoverished. In the present study the median quantity of LWD was 1 piece $100m^{-2}$. This result can be compared to North American studies on streams with pristine conditions where the measured density of LWD $100m^{-2}$ varied between 30 and 1700 (Bilby & Ward, 1989; Murphy & Koski, 1989; Fausch & Northcote, 1992; Ralph *et al.*, 1994; Flebbe & Dolloff, 1995). However, I do not know what the quantities of LWD of dead wood are in naturally dynamic benchmark ecosystems, nor the extent to which brown trout indicates other elements of biodiversity in small rivers.

5.6 Paper VI

To implement the EU Water Framework Directive's (Directive, 2000/60/EC) vision of reaching "*good ecological status*", actors and stakeholders at multiple levels representing different sectors involved with catchment management need to be equipped with appropriate knowledge as well as practical and reliable tools for on-the-ground policy implementation in social-ecological systems. Based on the ecological integrity concept and inspired by top-down hierarchical forest sector planning, and collaborative and communicative bottom-up participatory approaches (Fainstein, 2000; Lazdinis & Angelstam, 2004; Angelstam *et al.*, 2005b; Tress *et al.*, 2006; Angelstam & Elbakidze, 2007; Wu & Hobbs, 2007), I present a step-wise iterated approach to support the implementation of good ecological status of riverine landscapes. This is consistent with a proactive adaptive governance and management cycle (Mayers & Bass, 2004; Angelstam & Törnblom, 2004; Angelstam *et al.*, 2005b; Angelstam & Elbakidze, 2007) approaches that link policy, management, monitoring and assessment in iterated cycles within entire catchments. At the strategic level, regional quantitative gap analysis is a tool for assessing the extent to which ecological integrity is maintained by appropriate combination of protection, management and restoration of representative habitat types in an ecoregion. At the tactical level, a tool is Habitat Suitability Index (HSI) spatial modeling, which combines quantitative knowledge about focal species' requirements and the spatial distribution of resources into maps for visualization and scenario building. This allows both for assessment and subsequent spatial planning at different scales and time horizons. These two steps guide the management operations needed for the protection, maintenance and restoration at instream, riparian and landscape scales. There is, however, a scale mismatch between the need for this kind of systematic approach and reality. (Folke, 2002; Małgorzata, 2008) Thus monitoring programs and performance target need to be assessed, and tools for proper assessment, governance and management towards ecological integrity by various formal and informal organizations be developed.

Finally, adaptive governance and management need to be developed using participatory approaches that include relevant actors and stakeholders and enhance communication and collaboration. To fill knowledge gaps about performance targets and tools for governance applied interdisciplinary research is needed, which must become systematic in two dimensions. First, comparisons of reference landscapes with ecological integrity should be compared with altered systems to determine how much are enough of different structures and processes to secure viable populations of species used as indicators, and thus operationalise the terms "*good ecological status*" and "*ecological integrity*". Second, the idea that analytic deliberation, nested institutions and institutional variety open up for continuous experimentation, learning and change in society needs to be evaluated by studies of local and regional governance arrangements' ability to deliver good ecological status (Malgorzata, 2008).

5.7 Sustainable landscapes and catchments?

The widely accepted definition of sustainable development is that it is a development that meets the needs of the present without compromising the ability of future generations to meet their own needs (WCED, 1987). Sustainable development as a concept has been criticized for undermining the substance in environmental issues by removing the focus from the ecological conditions in landscapes and catchments to more general and pluralistic conditions within the whole society that ends up in involving everything concerning politics and society. Values vary greatly in detail within and between cultures, actors from different sectors, as well as between academic disciplines (*e.g.*, between constructivist sociologists, neoclassical economists and ecologists) (Tisdell, 1988; Daly & Farley, 2004). The introduction of social values to sustainability goals implies a much more complex and contentious debate, and those focused on ecological impacts tend to strongly resist non-ecological interpretations. However, the concept of sustainable development in Sweden has changed from an ecological perspective to a more general focus on economical and social perspectives. This means that the connection between sustainable development and environmental politics has become weaker (Lundqvist & Carlsson, 2004). Sustainable development tends to be an "umbrella concept" for a holistic vision for a future welfare society and the emphasis is focused on the process of the framework involving economical development, social welfare, co-operation, good environment, democracy and participation, health and equity in a general sense (Lundqvist & Carlsson, 2004). Ellen Wohl (2001) put out the question if it is feasible to save a species without saving the ecosystem in which that species evolved, in the context of the historical degradation of rivers and associated restoration efforts. Törnblom & Angelstam (in press) put out the question if it is feasible to talk about sustainable development when society at the same time is loosing vital ecosystems and viable populations.

5.8 Towards systematic assessments of ecological integrity

Today, no ecosystems are completely free from human impact. However, the degree of alteration varies widely. Sparsely settled mountain areas give an impression of wild, untouched, and unchanging nature. Yet, in many cases Mountain Rivers that appear to be pristine natural systems actually have been impaired as a result of historical human activities (Harding *et al.*, 1998; Wohl, 2001). Also, Wohl (2001) underscores the importance of distinguishing between the form, or the physical appearance, of a river and its function, which encompasses physical and chemical processes as well as biological communities associated to the river. Land-use patterns affect both the form and the function of the rivers throughout the world, yet these effects are still little recognized or understood.

A major problem in assessing the impact of catchment land-use on aquatic ecosystems is often the lack of reliable reference conditions that represent ecological integrity (Hynes, 1970; Liljaniemi *et al.*, 2002; Stoddard *et al.*, 2006), and appropriate performance targets based on the contents of legislative mandates and policy documents as the European Water Framework Directive (Stoddard *et al.*, 2006). In Western Europe and North America it is difficult to find catchments in such a natural state that they could be used as a reference for comparisons with variously affected catchments (Benke, 1990; Zwick, 1992; Muhar *et al.*, 1995). In the absence of totally natural benchmarks, one can use landscapes characterized by near-natural conditions as the best available reference or as described by Stoddard *et al.* (2006) as reference conditions for biological integrity. Such catchments can be found in remote areas of the former Soviet Union (*e.g.*, Kola Peninsula, Kamchatka) and in Canada and Alaska. These reference areas and catchments can be used to build systematic suites of landscape-scale and watershed-scale Habitat Suitability Index models (*e.g.*, Verner *et al.*, 1986; Scott *et al.*, 2002). The resulting thematic maps provide management guidelines to identify different variables, species and parameter values for the assessment, planning and management of the habitat networks at different spatial and temporal scales.

6 Further research needs

6.1 Focal species as indicators and communication tools

The studies presented in this thesis suggest that management of catchments' land use and cover, riparian composition as well as in-stream structures such as large woody debris (LWD), are needed for future conservation and restoration efforts towards ecological integrity. This conclusion is by no means new, but this thesis hopefully provides a new approach towards the understanding of ecological integrity of streams. Evidence was collected and presented supporting the hypothesis that landscape characteristics, relative to instream and riparian features, are far more important for benthic assemblages than previously understood. It is also evident that other aspects of stream ecosystems must be interpreted in a landscape perspective. This is definitely significant new information that, in turn, leads to new challenges. Streams have to be monitored in a broader and probably more comprehensive way.

From a management perspective, the suggestions on how to implement the landscape approach into the EU Water Framework Directive in Sweden and working procedures as well as methods have not yet been determined. This opens for the introduction of new knowledge, and new ways of producing knowledge, into the governance and management processes. Particularly interesting is the knowledge which is science-based but still practical enough for direct implementation in the management. In this way this thesis will be useful and beneficial for the development of future stream management. This thesis focuses on stream macroinvertebrate composition and distribution, viable freshwater pearl mussel populations, brown trout abundance in multiple spatial and temporal scales from the headwaters to the sea. I encourage others to replicate this approach for other species, and evaluate their functionality as focal species.

While previous studies have for some time pointed out the importance of in-stream conditions, riparian composition and landscape scale factors, still, very few studies have provided performance targets or new insights concerning thresholds for "how much is enough" of habitat within the different scales of a riverine landscape or a catchment. It is also necessary to evaluate the processes that affect the composition and structure of ecosystems (Noss, 1990; Angelstam & Kuuluvainen, 2004). This is stressed by the concept of ecological integrity (Piementel *et al.*, 2000; Norton, 2003). According to Karr & Chu (1999) biological integrity applies to sites at one end of a continuum of human influence, i.e. those supporting a biota that is the product of evolutionary and biogeographic processes. Frey (1977) described biological integrity as a "community of organisms having a species composition, diversity and functional organization comparable to those of natural habitats within a region". Adopting integrity as a management goal means aiming for a system that resembles this evolved state as much as possible (Angermeier, 1997).

Selecting a suite of focal species that can be used to derive performance targets at multiple spatial scales is a major challenge. Using the Baltic Sea region as an example, policies such as EC Habitat Directive (Anon., 1992) represent a first attempt of selecting prospective focal species for assessment of habitat networks. The species mentioned in the Annex 2 of the Directive and in the Natura 2000 network shall be the subject of special conservation measures concerning their habitat in order to ensure their survival and reproduction (*e.g.*, Cederberg & Löfroth, 2000). A first step in the selection process for riparian and aquatic systems could be to exclude from the species listed: 1) species that are dependent on other landscapes than rivers, streams, lakes, ponds, oxbows or riparian zones, and 2) species that could be hard to recognize or by deficits in knowledge, 3) species that have area requirements and use complex mosaic landscapes that are difficult to describe using simple land cover data.

Secondly, it should be checked whether or not the species selected from the EC Habitat Directive provide good coverage for the different spatial scales within a watershed and the ecoregions in the Baltic Sea region. Here, already collected monitoring data could be used to compare fish populations in different stream orders with associated land cover data sets covering tree species composition, age distribution, forestry intensity, fragmentation effects of riparian zones and streams. To test the hypothesis that there are non-linear responses of species in relation to structures and processes, several types of data sets could be used. These include (1) studies of different stream orders within catchments in the same biogeographical region, (2) evaluate riverine landscape reference conditions for catchments, and (3) combine existing monitoring programmes with the Swedish Electrofishing RegiSter (SERS) database of the National Board of Fisheries. The ambition would be to propose and evaluate the usefulness of potential indicator species identified in step two for different stream orders (spatial scales) within catchments that could reduce the costs for expensive and ineffective monitoring programs in aquatic environments.

Knowing what species require at multiple spatial and temporal scales is a necessary but not sufficient criterion for developing a systematic approach towards restoring the integrity of riverine landscapes. In addition, the knowledge must be in the right place, with the actors exercising governance as well as planners and managers. Communication with and among forest and river managers on the complex and often abstract criteria for selection of individual areas or stream reaches to be part of a functional habitat network could be alleviated if the principles were dressed in simple words (Uliczka *et al.*, 2004). A set of specialized species and their habitat requirements should therefore be scientifically evaluated before any recommendations as an operative planning tool. For example, many animals range over spatial scales compatible with those of forest and river management. In particular, fishes and mammals represent two well-studied taxonomic groups of animals. Many fishes and mammals are also well known by managers and some even function as flagship species, i.e. species useful for stimulating public interest in conservation (Simberloff, 1998) and sustainable use.

6.2 Linking ecosystems and institutions

Ideally, a combined terrestrial and aquatic ("TerrAquatic") approach should help societal actors and stakeholders to manage watersheds and landscapes in a more cost-effective way by adapting both terrestrial and aquatic perspectives at the same time, instead of today's scenarios with many separating activities of different sectors and actors that overlap in a geographical area. A TerrAquatic Gap Analysis (TAGA) approach should also evaluate monitoring programs, contribute to novel knowledge production and propose new perspectives and tools for governance, management and assessment of ecological integrity by various formal and informal institutions at multiple temporal and spatial scales (Angelstam *et al.*, 2003b; Lazdinis & Angelstam, 2004). There is, thus, an obvious need for transdisciplinary work in the context of (1) cycles of iterated environmental assessment of the degree to which short-term precondition for program success, and (2) policy cycles reflecting the long-term goals (generation objectives).

Such short- and long-term cycles of assessment requires integration of: (i) statistical methodology to increase the efficiency in the gathering of data, (ii) statistical methodology to analyse data, for example to separate variation due to human impact in nature from variation due to natural fluctuations, (iii) knowledge regarding obstacles and possibilities regarding communication between actors at different levels and for different kinds of institutions (formal and informal) involved in the implementation of measures in order to achieve the national environmental goals, (iv) way of analyzing how policies and guidelines are implemented, and (v) devising methods to integrate different dimensions of ecological sustainability.

A major challenge for applied research is to develop a balance between (1) how to translate to planners and managers results of assessments using narratives that convey the status and trend of a particular variable, and (2) how much simplification is possible within data and the messenger before losing the messengers expert credibility. I presume that this balance is different for different types of actors that are assumed to act upon the monitoring results (Angelstam & Törnblom, 2004).

Empirical support exists for the effects of critical thresholds in habitat abundance on animal populations in terrestrial landscapes (Andrén, 1994; Angelstam *et al.*, 2004). Similar thresholds seem to occur in aquatic landscapes, such as for dead wood and brown trout (Degerman *et al.*, 2004), or the percentage land cover of a certain land use within catchments for Plecoptera (Paper III). So far, however, there are too few studies available to support the formulation of performance targets for the conservation of aquatic biodiversity.

6.3 Conclusions

Based on the ecological integrity concept, and inspired by top-down hierarchical forest sector planning on the one hand, and collaborative and communicative bottom-up participatory approaches on the other, this thesis proposes a step-wise iterated approach to support the implementation of policies about good ecological status of riverine landscapes. This is consistent with proactive adaptive governance and management cycle approaches that link policy, management, monitoring and assessment in iterated cycles within entire catchments. At the first strategic level, regional quantitative gap analysis is a tool for assessing the extent to which ecological integrity is maintained by appropriate combination of protection, management and restoration of representative habitat types in an ecoregion. A second tactical level tool is Habitat Suitability Index (HSI) spatial modeling that combine quantitative knowledge about focal species' requirements and the spatial distribution of resources into maps for visualization and scenario building. This allows both for assessment and subsequent spatial planning at different scales and time horizons. These two steps guide the management operations needed for the protection, maintenance and restoration at instream, riparian and landscape scales. There is, however, a scale mismatch between the need for this kind of systematic approach and reality. Thus monitoring programs and performance target need to be assessed, and tools for proper assessment, governance and management towards ecological integrity by various formal and informal organizations be developed.

Finally, adaptive governance and management need to be developed using participatory approaches that include relevant actors and stakeholders to become understood and accepted. This requires enhanced communication and collaboration. To fill knowledge gaps about performance targets and tools for governance applied interdisciplinary research is needed, which must become systematic in two dimensions. First, comparisons of reference landscapes with ecological integrity should be compared with altered systems to determine how much are enough of different structures and processes to secure viable populations of species used as indicators, and thus operationalise the terms "good ecological status" and "ecological integrity". Second, the idea that analytic deliberation, nested institutions and institutional variety open up for continuous experimentation, learning and change in society needs to be evaluated by studies of local and regional governance arrangements' ability to deliver good ecological status. Even if it is difficult, time consuming and sometimes unrewarding, I hope this thesis will encourage others to employ a transdisciplinary approach!

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