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Emerging Concepts for Integrating Human and Environmental Water Needs in River Basin Management

Geoff Petts and Robert Kennedy

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Emerging Concepts for Integrating Human and Environmental Water Needs in River Basin Management

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ABSTRACT: The key to successful water and river management is the advancement of holistic approaches that seek to benefit human societies by sustaining the full range of resources created by rivers, including both physical and ecological services. This report describes the results of discussions held at the University of Birmingham, UK, during which participants sought to fill the conceptual gap that exists among water resource planners, flood engineers, and ecologists. Participants, including experts from Europe and the U.S. Army Corps of Engineers, attempted to advance and integrate concepts related to reference systems and sustainability and related to fully integrated water resource management within and between river basins. In a context of increasing pressures on (a) water supplies, wastewater treatment, and needs for flood management, (b) agricultural and forestry production systems, (c) land for urban expansion, and (d) nature conservation, recreation, and landscape restoration, participants discussed the primary challenge of managing changing rivers (changing flows, mobile sediments, and moving channels) in a diverse, dynamic, and highly connected system.

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Preface

Rivers and river corridors in Europe, North America, and other developed regions have been significantly altered over the past centuries. Riparian wetlands have been drained for development, stream and river channels have been straightened and confined within complex levee systems to prevent flooding, dams that alter natural hydrographic responses have been constructed to regulate river flows, and sediments have been routinely dredged as a means to maintain navigation channels. Many rivers and streams are now isolated from their floodplains, and native floodplain communities are threatened or lost. Trophic webs are dominated by anthropogenic sources of organic matter and are frequently impacted by contaminants. Effective management strategies must be implemented if these valuable resources are to persist.

However, society will continue to demand goods and services from riverine ecosystems. Given this reality, a return to pristine conditions is an unrealistic management goal, and new approaches to management must be sought. Such approaches must balance competing demands on the resource in ways that ensure its sustainability and persistence on the landscape. New and integrative concepts to guide such management efforts will be required.

This report documents the discussions and interactions of a select group of scientists, engineers, and managers during a workshop convened at the University of Birmingham, United Kingdom, on 16–18 September 2003. Participants, all of whom are involved in water resource management endeavors in the United States and Europe, shared experiences and knowledge bases and were challenged to explore concepts that could underpin the development and implementation of effective management strategies.

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The workshop was organized by Professor Geoffrey Petts, University of Birmingham, and sponsored by the European Research Office (ERO), U.S. Army Engineer Research and Development Center (ERDC), U.S. Army Corps of Engineers, London, with funding from the Water Operations Technical Support (WOTS) Program, ERDC. Dr. Robert Kennedy was Director, ERO. Mr. Robert Gunkel was Program Manager, WOTS.

At the time of publication of this report, the Commander and Executive Director of ERDC was COL James R. Rowen, EN, and Director was Dr. James R. Houston.

1 Introduction

Management Context

Historically, approaches in water resources engineering have been based on concepts developed to support societal needs to manage water quantity to meet consumptive demands, reduce flood damage, improve navigation, or diminish the impacts of waste loads. However, recent concerns for the continuing degradation of the environmental quality of streams, rivers, and estuaries and their associated wetlands raise difficult questions about current management strategies, especially as they often address water resource, water quality, flood management, and ecosystem conservation issues independently and over different scales of space and time. Linking concepts are clearly required to attain sustainable river ecosystems in the face of continuing intensification of water resource demands and predictions about climate change.

The urgent goal for water and river managers is to improve use efficiency and sustainable resource development. On the one hand, sustainable water resource management is needed to avoid the severe water shortage that is forecast in many countries for 2020 and beyond, with impacts upon water and food supplies, public health, and, potentially, national security. On the other hand, water is needed to sustain riverine ecosystems because they play a critical role in maintaining the ecological balance of the planet and provide a wide range of natural services for human societies.

This Water Operations Technical Support (WOTS) Workshop on emerging concepts in river basin management builds on the work of the Scientific Committee on Water Research (SCOWAR[1]) of the International Council for Science (ICSU). SCOWAR concluded (Naiman et al. 2002):

> *"the major challenge to freshwater management is to place water resource development within the context of fundamental ecological principles in order to maintain the ecological vitality (i.e., goods and services) of the system."*

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[1] The International SCOWAR Committee comprised Prof. S. E. Bunn (Australia), Prof. R. J. Naiman (USA, Chair), Prof. C. Nilsson (Sweden), Prof. G. E. Petts (UK), Prof. G. Pinay (France).

It also builds on three recent Symposia: *Remedial Strategies in Regulated Rivers*, Lycksele, Sweden, 1995; *New Approaches to River Management*, Nijmegen, The Netherlands, 1998; and *River Restoration in Europe*, Wageningen, The Netherlands, 2000.

In river management there is a slow shift from the old paradigm of sustained yield to one that embraces ecosystem management and collaborative decision making, promoting consensus building among stakeholders. This workshop focuses on the first, namely the advancement of concepts and principles that embrace ecosystem management in water resources planning.

Ecosystem management, incorporating the conservation of biodiversity and the protection of the full range of ecosystem functions and services, needs to be seen as a tool for development, not as a constraint to development. This is important also because many argue that, while new concepts redefine problems and focus on new questions, it is the advancement of new tools that achieves new knowledge and, then, better management.

Ecological Context

The fundamental structural relationships of the ‘fluvial hydrosystem’ (sensu Amoros and Petts (1993) in French; Petts and Amoros (1996) in English) are well established. A river network comprises a nested hierarchy of spatially connected units (sectors>landforms>mesohabitats), each having different levels of sensitivity and recovery; the whole being subject to climatically driven temporal variations in physico-chemical processes (flows, sediment loads, water quality, biological resources), modified by the downstream sequencing of storage, utilization, transformation, and erosion-release processes and by the temporal and spatial dynamics of biological populations and their interactions.

From headwaters to mouth, patterns of physical habitat organization give a river network a more or less typical structure dominated by (a) the catchment-scale sequence of (1) altitudinal and latitudinal temperature gradients and longitudinal changes in organic matter sources and trophic webs, superimposed upon (2) sector-scale channel styles interrupted and reset by (3) transition zones; and (b) dynamic mesohabitat mosaics that (1) are typical of individual landforms and relate to patterns of plant and animal distribution and (2) form successional sequences that are interrupted and reset by disturbance events so that each mesohabitat is represented by a more or less complete sequence of successional stages depending upon the magnitude and frequency of disturbance events (floods–droughts) and the succession–recovery rate.

It has been suggested (Naiman et al. 2002) that there are three overarching ecological principles for water resources management. These are deceptively simple but have been distilled from decades of research in river ecology.

a. The natural flow regime shapes the evolution of aquatic biota and ecological processes.

b. Every river has a characteristic flow regime and associated biotic community.

c. Aquatic ecosystems are topographically unique in occupying the lowest position of the landscape, thereby integrating catchment-scale processes.

These have been expanded into another six key principles by Bunn and Arthington (2002), Nilsson and Svedmark (2002), and Pinay et al. (2002):

a. Flow is a major determinant of physical habitat, which in turn is a major determinant of biotic composition.

b. Aquatic species have evolved life history strategies primarily in direct response to the natural flow regime.

c. Maintenance of natural patterns of longitudinal and lateral connectivity is essential to the viability of populations of many riverine species.

d. The duration and timing of inundation of alluvial soils has a major influence on biogeochemical processes, especially organic matter decomposition and nitrogen cycling.

e. The riparian zone is a transition zone between land and water and is plant species rich when compared to surrounding ecosystems.

f. The invasion and success of exotic and introduced species in rivers is facilitated by the alteration of flow regimes.

The European Dimension

This WOTS Workshop seeks to provide a European perspective; one that places ‘sustainable water management’ within the context of *naturalized* rivers—those where the morphological and ecological configurations are compatible with the magnitudes and rates of physico-chemical processes driven by the contemporary catchment ecosystem (Petts et al. 2000). This includes:

a. A long history of human-induced environmental change.

b. Managing for sustainability in densely populated, urban and industrialized, catchments.

c. A strong drive toward nature conservation and landscape restoration.

There are increasing pressures on (a) water supplies, wastewater treatment, and needs for flood management, (b) agricultural and forestry production systems, (c) land for urban expansion, and (d) nature conservation and recreation and landscape restoration. The 21st Century catchment may show the following:

a. Complex land use changes (accelerated soil erosion leading to high loads of fine sediments in catchments subjected to high-density grazing or high-intensity cultivation; reduced sediment yields because of reservoir impoundment; and flow regulation and abstraction from headwater [source] catchments) and rates of land use change that are much faster and much more dynamic than in natural systems.

b. Channel dynamics limited to a beads-on-a-string model within a fluvial corridor: resetting of mesohabitats is limited to narrow strips and occasional ‘islands’ (large patches of ca. 10,000 m^2) sustained by contemporary processes and bounded by ‘managed’ floodplain and terraces.

c. Trophic webs dominated by anthropogenic (including agricultural and domestic) sources of organic matter and nutrients, including carbon, nitrogen, and phosphorus.

However, considerable research gaps remain to be filled before analytical approaches can be developed to determine the morphological and ecological configurations that sustain naturally functioning ecological systems. Adaptive management constantly seeks to improve and adjust as understanding of how ecosystems work advances. But do we monitor the most appropriate indicators? Should management be more proactive in advancing experimentation, taking actions in order to learn more about the managed system?

2 Role of Landscape Ecology in River Management

Concepts in Landscape Ecology

In the Netherlands, landscapes are studied by pattern (the geographical approach) and by process (the ecological approach). There is not much progress in the development of a deterministic basis for holistic landscape ecology. The synthesis of the ecological approach and the geographical approach has been slow to occur, in part owing to the lack of theory dealing with spatial patterns in landscape ecology. Recent developments in metapopulation theory, theories about habitat fragmentation and the relation between landscape patterns and biodiversity, have triggered the interest of ecologists in landscape level approaches. Questions of spatial and temporal patterns in ecology, and more specifically the roles of space in population dynamics and interspecific interactions, have recently led to a new discipline called spatial ecology. Principles of spatial ecology are applicable in landscape ecology (Wiens 1997, Opdam et al. 2002). According to Tockner et al. (2002), a landscape ecology approach holds the potential for developing a truly holistic perspective of river corridors, one that rigorously integrates structure, dynamics, and function.

A number of fundamental concepts that underpin current scientific knowledge of riverine ecosystems have been formulated over the past 25 years. The river continuum concept (RCC) (Vannote et al. 1980) and the flood pulse concept (FPC) (Junk et al. 1989) are the best known and most comprehensive concepts. The RCC offers a framework for characterizing pristine running water ecosystems, describing the structure and function of communities along a river system, from its source to mouth. The FPC has made a major contribution to our understanding of river–floodplain interactions. The concept is based mainly on large, tropical lowland rivers, and Junk et al. (1989) term the floodplain area influenced by the predictable flood pulse, the aquatic/terrestrial transition zone (ATTZ). This area is periodically inundated by the lateral overflow of rivers and lakes or by direct precipitation or groundwater. Tockner et al. (2000) extended the concept to temperate areas by including information derived from glacial and lowland floodplains. They studied functional processes at low discharges *(contraction phase)* and high discharges *(expansion phase)*, and their consequences on habitat heterogeneity. A key concept integrating geography and ecology in river studies is *connectivity*. Connectivity is defined as the strength of interactions across ecotones (i.e., transition zones between adjacent patches). Ecotones, connectivity, and succession play major roles structuring the spatio-temporal heterogeneity, leading to the high biodiversity that characterizes floodplain rivers.

Landscape Connectivity

Landscape connectivity consists of a structural component and a functional (biological) component. Traditional measures of connectivity in landscape ecology have focused on structural components, such as nearest-neighbor distances, patch area, edge-to-area ratio, etc. The functional component focuses on the response of individuals to landscape features (behavior) and the patterns of gene flow that result from those individual responses. In population ecology, connectivity is typically measured with biological criteria, including mark–recapture, and measures of genetic structure. Integrative indices of connectivity are nonexistent, but Brooks (2003) argues that such an index can emerge only from the explicit comparison of the temporal and spatial scales of the structural and functional (biological) components of connectivity.

Structural components

The structural scale of landscape connectivity can be assessed through a combination of graph theory applications, geographic information systems (GIS), and remote sensing (RS) technology. Bunn et al. (2000) suggest that graph theory can be used in applications concerned with landscape connectivity. Graph theory is a well established branch of information technology. It is a rather classic theoretical modeling approach, using algorithms on mathematical graphs. It employs fast algorithms and compact data structures that can be easily adapted from the habitat level to landscape level. Graph theory is the study of connectivity in stochastically generated structures, and has been the basis of neutral models in landscape ecology. Neutral landscape models predict a nonlinear threshold response of a landscape to habitat fragmentation. Above the threshold value, habitat destruction simply results in a loss of suitable habitat, but at the threshold even small losses of habitat result in the rapid breakup of the landscape into disconnected clusters. Graphs are commonly used to maximize connectivity in road, telephone, and computer networks. These graphs are composed of points (nodes) and lines (edges) that are used to represent patches and connections between them, respectively. Both edges and nodes can be given weights that can represent structural characteristics, such as nearest-neighbor distance, or biological characteristics, such as the cost of moving between patches. Bunn et al. (2000) showed the example of two animal species that share the same habitat but have different dispersal capabilities. Graphs using GIS overlays to define habitat patches were constructed, and the functional distance between the patches was determined. Using graph operations, they found that the landscape is fundamentally connected for species I and fundamentally unconnected for species II. Without denying the strength of graph theory in this example, one view is that an ecologist, using his intuitive, qualitative knowledge, might come to a similar conclusion. The advantage of the graph–theoretical approach, however, over other modeling techniques is that it is a heuristic framework that can be applied with very few data.

Satellite remote sensing (RS) and geographic information systems (GIS) are emerging technologies in environmental sciences. They offer the opportunity to gain insight in interrelationships of scale, pattern, and process, a paradigm that has reached momentum in the fields of biogeography and landscape ecology (Walsh et al. 1998). According to Kerr and Ostrovsky (2003), RS provides the only means of measuring the characteristics of habitats across broad areas, connecting to environmental changes that occur as a result of human or natural processes (e.g., climate change). There is, however, the problem of scale mismatch between traditional field ecological data and most remote sensing data sources. Studies in the field provide detailed measurements over small areas at different times, whereas the most commonly used remote sensing data provide synchronous measurements over broad areas but with reduced potential for local detail.

A complementary structural approach to connectivity is derived from fractal geometry. It is increasingly recognized that natural patterns often show very irregular patterns that can only be very roughly characterized by methods of Euclidean geometry. A classic example is the question how long a specific shoreline along a river is, a question to which the answer is of course relevant for any wader bird that uses it as a breeding and foraging habitat. The answer depends on the length of the ruler with which we measure this length. If we define the total length as the product of the length of the ruler and the number of times we need to flip it over to measure the shoreline, we get an ever-longer length when using a shorter ruler. When the log of the length of the ruler is plotted against the log of the measured length with that ruler, we usually get a straight line. This indicates that shorelines as a natural shape cannot be characterized by integer dimensions in Euclidean geometry, but by fractions of a dimension or a ‘fractal dimension’ in so-called fractal geometry. According to Olff and Ritchie (2002), fractal geometry seems to be a promising approach for linking population and community processes to landscape spatial structure.

Functional components

Habitat loss is probably the most important factor causing species declines worldwide. Thus, it is crucial to understand the mechanisms that underlie the effects of habitat loss. Because this process often involves habitat fragmentation, metapopulation and landscape concepts that focus on spatial effects play a major role in studies of habitat loss. A major paradigm for studying the ecology of habitat loss and fragmentation is the metapopulation view, which states that metapopulation persistence depends on the interplay between extinction from occupied patches and recolonization of empty patches. Simple metapopulation models are deterministic, although nature is stochastic (Sih et al. 2000).

Because it is impossible to monitor and manage every aspect of biodiversity, several shortcuts have been proposed whereby single species are protected and monitored. A number of concepts are currently in use. The *indicator species* concept is problematic because there is no consensus on what the indicator is supposed to indicate and because it is difficult to know which indicator species is the best, even when there is agreement on what it should indicate. The *umbrella species* concept is dealing with species that need such large tracts of habitat that saving them will automatically save many other species. The concept of the *flagship species*, normally a charismatic large vertebrate, is one that can be used to anchor a conservation campaign because it arouses public interest and sympathy (e.g., the beaver, *Castor fiber*, in lowland river stretches of Western Europe). Management of *keystone species* may combine some attractive features of single-species management and ecosystem management. If the keystone affects many other species in its community, and hence is functionally crucial to a suite of other species, its management may maintain them. Ecosystem management is a suggested solution to the problems posed by single-species management focused on indicator, umbrella, flagship, and keystone species. The key feature in ecosystem management is a focus on ecological processes rather than individual species (Simberloff 1998).

Linking patterns and processes in landscape connectivity means combining the structural and functional components emergent in landscapes. Ecological processes are essentially stochastic. Spatial stochastic models play an important role in understanding and predicting the behavior of complex systems. Such models may be implemented with explicit knowledge of only a limited number of parameters relating to spatial relationships among locations. Consequently, they are often used instead of deterministic–mechanistic models, which may potentially require an unrealistically large number of parameters. Methods to quantify aspects of spatial patterns that can be correlated with ecological processes are classified as landscape pattern indices (LPIs). The computations of landscape metrics have been facilitated by software developments. Huge amounts of data can be summarized in a single number, without prior knowledge of a landscape and its processes and organisms.

LPIs include the shape of patches, edges of pixels, or focus on diversity. Once a stochastic process is defined and then modeled, in the form of a computer code, any number of realizations can be generated. For example, starting from a Landsat image-based forest cover classification database (pixel size = 30 m), a series of simulated landscapes can be derived (Fortin et al. 2003, Tischendorf 2001).

Landscape Ecology and River Management in a European Context

Integration of structural and functional components in landscape connectivity is a scientific challenge. But there is more in river science. The history of rivers and streams is as much a social and technological history as it is a scientific one. Rivers are the lifeblood of nations, and the control of their waters has been fundamental to the building of human civilizations (Petts 2001). Evaluating landscape change requires the integration of the natural and social sciences. Indicators of landscape health include indicators of integrity, measuring biological condition relative to the condition in landscapes largely unaffected by human activity, and indicators of societal values, based upon intergenerational concerns at regional scales, that govern changes in highly modified landscapes. The ‘legacy of evolution’ and the ‘legacy of culture’ require integration for effectively coping with environmental change (Rapport et al. 1998).

The second half of the 20th century saw a revolution in agricultural practice that surpassed any previous agricultural revolution. Economic and technological incentives to increase agricultural productivity in postwar Europe have resulted in unprecedented rapid agricultural intensification over the past 60 years, causing widespread declines in farmland biodiversity in recent decades. There is now much evidence to suggest that the decline in farmland biodiversity is related to changing farming practices.

The most impacted riparian corridors with respect to land use are found in Europe (catchments with population densities of more than 200 people per km^2), where about 60 to 95 percent of the entire riparian corridor has been transformed to cropland or is urbanized (Tockner and Stanford 2002). In other words, the river landscape in Europe is an agricultural landscape. The largest decline of any wetland category has been of forested freshwater wetlands, primarily riverine floodplains. Originally, floodplain forests along the lowland section of the River Rhine covered 60 percent of the surface area of the floodplains; nowadays, 70 percent of that area is cultivated and transformed into agricultural land, and forest cover is less than 5 percent (Nienhuis et al. 2002). We have to realize that European floodplains are cultural floodplains, where the ‘legacy of culture’ weighs heavily.

The European floodplains are, at their best, seminatural landscapes, and attempts to ‘restore’ these landscapes are directly confronted with the question: what is the target situation, which ‘leitbild’ is applied? Conservation and restoration schemes refer too often to the ‘natural’ situation. But this ‘natural’ river is an echo from the past. River management should strive to optimize ecosystem integrity in regulated river basins. Ecosystem integrity refers to the maintenance of the community structure and function characteristic of that particular ecosystem, together with the capability of the system to support services and goods to humans (paraphrasing De Leo and Levin 1997). The integrity of a seminatural river floodplain in Western Europe should be recognized, where, owing to the management strategy of generations of farmers, new landscape elements, comprising new biodiversity, have been added to the original ‘natural’ landscape. Protection and restoration targets of lowland river floodplains should focus on the small-scale agricultural management practices of the past: everywhere a different way of management, but in a sustainable way (i.e., constant over time), in contrast to the modern large-scale and intensive agricultural practice: everywhere the same way of management, but in a nonsustainable way (i.e., rapidly changing over time).

The European cultural river-landscape is characterized by large homogeneous patches of intensively fertilized and cultivated land. This picture is not uniform for the entire European Union, especially with the incoming countries in 2004. In broad lines, going from west to east, the trend is from the overdeveloped to the underdeveloped countries, from the heavily regulated and urbanized rivers, where many habitats have been lost or fragmented, to the less regulated and less urbanized rivers, where stretches of near-natural and seminatural river habitat are still present. There is a delicate balance between ecological and economic interests that is characterizing sustainable river management. In Western European countries, the scales were tipped after World War II in favor of economic interests. Recently, however, there has been a reverse movement in favor of ecological interests: millions of Euros are spent to rehabilitate degraded rivers, and particularly to enhance water quality. In contrast, in Central European countries the economy is growing, leading to changes in land use, the intensification of agricultural practice and increasing pressure on river systems at the expense of ecological values (Nienhuis et al. 2000).

A large part of the European legislation on environmental affairs is already centralized in Brussels. In principle, Europe can avoid the squandering of ecological values in the Central European countries. The scale may still be turned in favor of ecological interests by avoiding the expensive mistakes made in Western Europe. This can be done by preserving the near-natural and seminatural stretches of river beds and by accommodating economic interests in a sustainable way.

The patch dynamics approach, the nested scalar approach, from the microhabitat to the catchment level, is widely accepted, both in the United States and Europe. However, the monitoring and assessment procedures to establish water quality or to quantify the overarching ecosystem integrity differ widely among nations. England, France, Germany, the Netherlands, and other countries have all developed their own protocols and survey methodologies. It is strongly advised to strive after uniformity in the European Union, and the European Water Framework Directive (EWFD) should be used as a means of exerting pressure to reach that goal. The EWFD is aiming at ‘good ecosystem quality’ for river catchments within 10 to 15 years; however, as long as the methodologies to measure ‘ecosystem quality’ are not standardized, it will be impossible to compare the attempts of the European countries to reach the common goal (Zalewski 2002).

3 Linking Pattern and Process Along River Corridors

Introduction

One of the most challenging topics in ecology is the development of principles for guiding the restoration of aquatic and terrestrial systems and developing methods to assess the success of restoration projects. Riparian ecosystems are particularly distinct systems because of their open link to adjacent ecosystems, their interface position between land and water, and the constraints that hydrological and morphologic dynamics place on their flora and fauna. Most riparian ecosystems are also topographically unique systems, occupying the lowest position in the landscape, thereby integrating upstream catchment-scale processes.

Globally, riparian ecosystems are key strategic natural resources, which in the future will play a pivotal role as focal nodes for biodiversity and human development. In Europe and Japan, more than 50 percent of the entire population lives on former floodplains. In the developing world, the combination of rapid increase in human population density, high urbanization rate, and economic development will lead to major pressures on riparian ecosystems, primarily by altering the natural flow regime.

Riparian Corridors: Focal Points of Biodiversity

Their highly dynamic nature makes riparian ecosystems among the most biologically productive and diverse systems on earth (Naiman et al. 1993, Tockner and Stanford 2002). Indeed, far more species of plants and animals occur in riparian ecosystems than in any other landscape unit in most regions of the world. In the Pacific coastal ecoregion of the United States, for example, approximately 29 percent of wildlife species found in riparian forests are riparian obligates (ranging from 12 percent of mammals to 60 percent of amphibians). Although less than 1 percent of the landscape of the western United States supports riparian vegetation, this vegetation provides habitat for more species of breeding birds than any other vegetation association (Knopf and Samson 1994).

These riparian zones are, however, one of the least investigated ecosystems in terms of their contribution to biodiversity.

There exist some basic principles that drive the ecology of riparian ecosystems (Bunn and Arthington 2002). The flow regime determines the ecological processes and the temporal patterns of variability in riparian communities. Fluvial dynamics, including the expansion and contraction of surface waters (*flood and flow pulses*), is also the driving force that sustains connectivity between floodplains and the river channel (Ward et al. 2002). Even small changes in the relative contribution of individual water sources may drastically alter species composition and diversity. For example, local groundwater upwelling is often associated with a higher-standing crop of algae, higher zoobenthos biomass, faster growth rates of cottonwood trees, and a higher species richness of woody and herbaceous plants (Harner and Stanford 2003). Despite its overwhelming importance in floodplains, hydrology is often given only cursory attention in restoration and mitigation projects (e.g., Bedford 1996).

Environmental flow requirements and management

In recent years there has been a major move toward the evaluation of river flow regimes in relation to the needs of natural ecosystems (both in-stream and on floodplains) as legitimate users of fresh water, next to other users, such as agriculture, industry, and domestic water supply (Naiman et al. 2002). Both high flows and low flows may be managed (in terms of timing, frequency, magnitude, and duration) to encourage sustainable river-floodplain ecosystems. Unfortunately, the amount of water allocated to rivers through environmental flows is rarely sufficient to replace the small to medium floods that regulation and abstraction have affected. In these situations, water is best targeted to key ecosystems such as Ramsar wetlands and seminatural floodplains. The major scientific challenge in the near future will be to understand the different modes of adaptation of the fauna and flora to specific flow regimes (e.g., Lytle and Poff 2004, see Table 1), to evaluate the flow requirements of species and ecosystem processes, and to integrate this knowledge into decision support models. The knowledge of flow requirements for vulnerable and endangered fish species, for example, is very limited, but this knowledge is a prerequisite for the protection of species and the use of these species as environmental indicators.

Structural and functional indicators

The selection of indicators of environmental conditions is crucial for the interpretation of environmental changes. These indicators need to be sensitive toward human impacts, ecologically meaningful, capable of being integrated over different spatial and temporal scales, and easily and economically applied. It is of prime importance to select indicators that mirror the major ecological functions,

Table 1
Major Disturbance Regime, Availability of Refugia, and Adaptation Strategies of Benthic Invertebrates Along a Riparian Corridor (major geomorphic sections) (after Tockner et al. in prep. c)

Geomorphic Type	Disturbance Regime	Refugia	Adaptations
Headwater (constrained)	Avalanches, debris flows	Tributaries, hyporheic zone, surface roughness	Drift (mobility), morphological adaptation.
Mid-section (braided)	Morphological dynamics, flow and flood pulses	Shore areas, dead zones, wood, hyporheic zone	Mobility, life history strategies, risk spreading.
Lowland (meandering)	Flooding, channel migration	Floodplain, woody debris, backwaters	Physiological/ethological adaptations, diapause.

directly or indirectly, along riparian corridors. The following paragraphs present three innovative indicators for assessing the ecological integrity of riparian corridors: (a) aquatic and terrestrial "islands" as structural indicators, (b) the availability of refugia to indirectly evaluate ecosystem resilience, and (c) the composition of floating organic debris as a functional indicator of longitudinal and lateral connectivity.

Aquatic and terrestrial "islands." Riparian corridors are characterized by a high diversity of aquatic and terrestrial habitats, including isolated water bodies and vegetated islands. Both isolated water bodies and islands are endangered landforms in Europe and elsewhere in the developed world. Vegetated islands are "high energy landforms" (Osterkamp 1998). Their formation requires a natural flood regime, an unconstrained river corridor, a sediment source, and a source of large woody debris, a combination of conditions not present in highly managed river systems (Ward et al. 2002, Gurnell and Petts 2002). For example, over 650 vegetated islands (> 0.007 ha) occur along the corridor of the Fiume Tagliamento, the only large morphologically intact Alpine river remaining in Central Europe (Tockner et al. 2003). Islands are, however, among the first landscape elements to disappear as a consequence of river regulation. For example, only six islands remain of the ca. 2,000 islands historically present in the Austrian Danube.

We suggest islands as ecosystem-level indicators of the environmental condition of a river corridor (Tockner et al. 2003). Ecologically, islands are pivotal landscape elements. They represent early successional stages, are colonized by a diverse and often endangered fauna and flora, are almost devoid of invasive species, have a high perimeter-to-area ratio, serve as stepping stones for migrating organisms such as small mammals, and serve as important natural retention structures along riparian corridors. Islands increase the shoreline and are therefore very important cover habitat for fish, distinctly enhancing the in-stream structure, which provides hiding places and refuge from predators. The presence of vegetated islands also controls the diversity of aquatic habitats (Arscott et al. 2000, Gurnell and Petts 2002). Recent investigations have shown that floodplain ponds, which are often closely related to vegetated islands, contribute disproportionately to aquatic diversity along river corridors. Although they only cover a small proportion of the total aquatic area (less than 5 percent in most cases), they contribute more than 50 percent of total species richness (Karaus et al. 2005). Again, regulation leads to the rapid elimination of the most sensitive habitats, such as concave and convex islands.

Availablity of refugia. Refugia are areas from which recolonization occurs following a disturbance event. The distribution and utilization of refugia are of critical importance for maintaining the ecological stability of ecosystems. Therefore, the potential availability of refugia can be used as an indicator of ecosystem resilience, which is the capacity of an ecosystem to respond to disturbance. As the dominant disturbance regime is changing along the river corridor ("disturbance cascades" sensu Montgomery 1999, see Table 1), the relative importance of individual refugia changes as well. Therefore, we suggest measuring the potential availability of refugia at three different scales: vertically as the permeability of bed-sediments, laterally as shoreline length/shore area (see Figure 1), and longitudinally as the relative proportion of unmodified tributaries (up to a distance of about 10 km, depending on stream size). All variables are easily measured in the field. Based on these variables, one can develop a standardized Functional Capacity Index (FCI):

Variable 1: Local refugia (interstitial):
Permeability of bed-sediments (V_{perm})

Variable 2: Local refugia (riparian area)
Relation: riparian area/channel width (V_{rip})

Variable 3: Regional refugia (V_{trib})
Number of tributaries, which are intact (hydrologically, morphologically) (up to 10 km upstream; V_{trib})

FCI = $(V_{\mathrm{perm}} + V_{\mathrm{rip}} + V_{\mathrm{trip}})/3$

Standarization (relative reference; 0–1): Pi' = $P_{\mathrm{i}} - P_{\mathrm{min}}/P_{\mathrm{max}} - P_{\mathrm{min}}$

FCIs have already been successfully applied in wetland assessments (e.g., Brinson and Reinhardt 1996). The variables used for calculating the FCI only measure the availability of refugia indirectly. It is, however, necessary to calibrate the index against standards obtained from reference ecosystems (see Role of Reference Ecosystems following). Again, some basic research is still required to develop and calibrate this index.

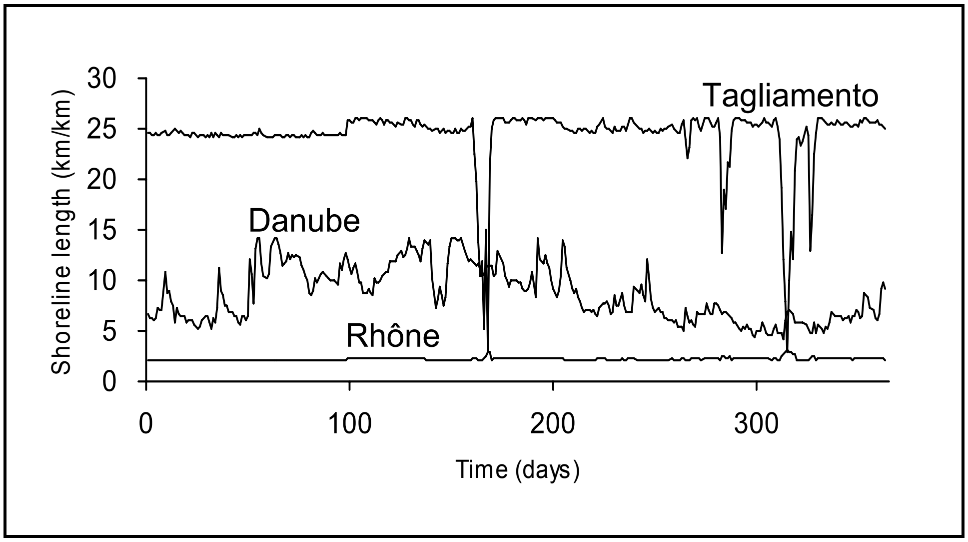

Figure 1. Shoreline length development (km per river-km) in three river-floodplain ecosystems over a 1-yr period (after Tockner and Stanford 2002)

Floating organic debris. Floating organic matter is transported at the water surface and is the least understood component of sediment transport in rivers. It links both aquatic with terrestrial compartments and upstream with downstream segments of river ecosystems (both energetically and as a vector for terrestrial and aquatic organisms). During flood events, large amounts of organic material and organisms float downstream. During the decreasing limb of the hydrograph, organic material aggregates and accumulates in "dead zones" and at retention structures along shoreline habitats. Eventually, it is deposited along shorelines, where it forms distinct "drift lines." With an increase in the water level, deposited material becomes resuspended and transported downstream (Figure 2).

Floating organic matter serves as a major dispersal vector for aquatic and terrestrial organisms along riverine corridors and is a cover extensively used by juvenile and adult fish. Recent results from the Tagliamento River demonstrate that organic debris was much more diverse at the surface compared to the water column (Langhans and Tockner, in prep.). Coarse organic matter particles, such as wood, fruits, and grass, were exclusively transported at the water surface. The abundance and composition of invertebrates change rapidly between transport, accumulation, and deposition phases. The number of organisms associated with floating organic debris was on average twenty times higher than that in the water column and was primarily composed of terrestrial organisms. Many of the organisms are transported over long distances (tens of kilometres). The removal of organic debris upstream of hydropower plants is considered to have a major impact on the ecology of river systems, leading to a significant decline in local riparian species richness (Andersson et al. 2000a, b).

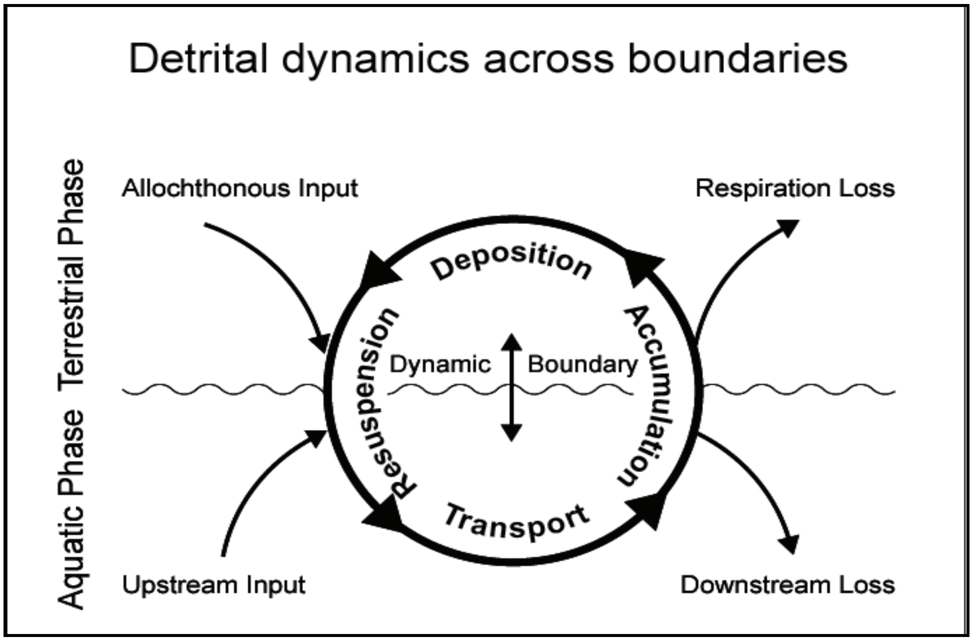

Figure 2. Floating organic matter dynamics across aquatic-terrestrial boundaries (after Langhans and Tockner, in prep.)

The elemental composition and the faunal and floral components of floating material can be used as an indicator of the integrity of entire river corridors. The main driving factor in the cycling of floating organic debris and its associated fauna is the pulsing of flow. However, basic research is required to establish and calibrate floating organic matter as an integrative indicator of connectivity along riparian corridors.

Role of Reference Ecosystems

A major problem associated with restoration and management schemes is the identification and definition of reference conditions. References can be defined geographically (e.g., Tagliamento for Alpine Rivers), historically (using historic information), or theoretically (using ecological principles). Many river management and restoration concepts fail because of the lack of fundamental knowledge of the structural and functional features of morphologically intact river corridors. Until quite recently, most concepts in river ecology were based on the implicit assumption that rivers are stable, single-thread channels hardly interactive with adjacent riparian zones and floodplains. Unfortunately, many European rivers are in such a state, but it must be recognized that this is not the natural condition. We believe that this incomplete understanding constrains scientific advances in river ecology and renders management and restoration initiatives less effective.

A list of the remaining areas that are mostly unexploited, and not stocked with fishes, can be a guide to opportunities for effective conservation. These are also those areas where we might conserve the widest range of biodiversity with a minimum of conflict, and they can also be seen as “seeds of wilderness.” Indeed, in Europe, North America, and Japan, the last remaining dynamic floodplains (such as the Tagliamento River in NE Italy, Figure 1) and free-flowing, morphologically intact river stretches represent the only remaining ecosystems in these regions where large-scale natural disturbance events still occur and where we can investigate the linkage between patterns and processes across different scales.

Conclusion

Riparian ecosystems are unique and dynamic systems that link rivers with their catchments. They are highly productive environments, supporting a diverse biota, but they are also intensively used by humans for agricultural and urban development, resulting in loss of biodiversity, low fish population density, and ecological functioning. As with most ecosystems under threat, our priority for floodplains is to conserve those that are still intact and to attempt to rehabilitate those that are degraded. In both cases, protecting or restoring key components of the natural flow regime is essential, while maintaining sustainable use of floodplain resources by local communities, particularly in developing countries. Finding this compromise between conservation and resource use requires a greater understanding of the role of flow in relation to other stressors to driving ecological processes in floodplains. Floodplain management and restoration strategies must also take into account climate change models that predict significant changes to flow regimes in most of the world’s rivers, especially in temperate and arid regions.

4 Environmental Flows

Introduction

Water resource developments involving flow regulation have a long history and at present are being undertaken on an unprecedented scale worldwide (Rosenberg et al. 2000). Almost 10 years ago, Dynesius and Nilsson (1994) calculated that 77 percent of the discharge of the 139 largest river systems in North America, Europe, and the former Soviet Union is strongly or moderately affected by flow-related fragmentation of river channels. Impacts on water resources include impoundments, river diversions, interbasin water transfers, and abstraction, both from surface waters and from groundwater reservoirs. Along with agriculture and domestic and industrial water supply, the hydropower industry is a major stakeholder of the world's water resources.

Water resources are finite, and the escalation of the demand for water resources has resulted in many conflicts, not only in regions where water is scarce, but throughout the world. These conflicts have prompted the realization that *environmental flows* are prerequisite for a sustainable utilization of water resources and a necessary component of any long-term water resources management strategy (see Special Issue of *River Research and Applications* [2003; 19, 5-6] on "Environmental Flows for River Systems").

The European Union Water Framework Directive, ratified in 2002, has been adopted by most European countries. The Directive has clearly defined environmental standards and states that the aim should be to achieve at least "good ecological status" in all surface waters and groundwater. Flow determines to a large extent the nature and development of the freshwater ecosystem (Petts and Maddock 1994) and thus is of crucial importance in maintaining ecological status.

Environmental Flows

River ecosystems, as well as housing a unique and diverse biota, provide ecosystem goods (e.g. drinking water, fish, electricity) and services (e.g. water purification, flood mitigation, recreation). Healthy rivers and associated ecosystems also have an intrinsic value in terms of cultural and aesthetic significance, although these may be difficult to quantify. The flow regime is one of the overriding determinants of the character of a river ecosystem, reflecting its geographical location and the geological and topographic features of the area (Statzner and Higler 1986). Recognition of the need to establish the extent to which the flow regime of a river can be altered from natural, for the purposes of water resource development and management, while maintaining structural and functional integrity, or an accepted level of degradation, of the ecosystem has provided the impetus for accelerated development of a relatively new science of environmental flow assessment (Tharme 2003).

Environmental flows are a requirement for maintenance of good river health and can be defined as: *the water regime provided within a river, wetland or estuary to maintain ecosystems and their benefits where there are competing water uses and where flows are regulated* (IUCN 2003). An environmental flow assessment for a river produces one or more descriptions of possible modified hydrological regimes for that river and the environmental flow requirements, each linked to a predetermined objective in terms of the ecosystem's future condition. These objectives may be directed, for instance, at the maintenance or enhancement of the entire river, including its various aquatic and riparian biota and components, maximizing the production of commercial fish species, or conserving particular endangered species, as well as protecting features of cultural or recreational interest. Typically, environmental flow assessments are performed for river systems that are already regulated, or the focus of a proposed water resource development such as hydropower.

The level of resolution varies from a single annual flow volume through to a comprehensive, modified flow regime where the overall volume of water allocated for environmental purposes is a combination of different monthly or even daily allocations. Poff and Ward (1990) emphasized the significance of disturbance in shaping ecological processes and patterns in rivers so flushing flows are frequently included. The scale at which the assessment is undertaken may also vary widely, from a whole catchment to a single river reach. Different methodologies are therefore necessary over such a broad range in spatial scale and levels of resolution. Other constraints include the time available for assessment, availability of data, technical capacity, and finances.

Principles governing environmental flows

The provision of environmental flows is not intended to mimic a pristine river. A regulated system, by definition, cannot reproduce all aspects of natural flows while providing for competing uses (Ward and Stanford 1983). There will always be a cost (Nilsson 1996)! Thus, a distinction is made between the amount of water needed to maintain an ecosystem in near pristine condition and what might eventually be allocated to it following a process of environmental, economic, and social assessment.

How much can then be taken out? This is not always a question of percentage, but how it varies over time. The challenge of proving environmental flows is in part determining which elements are critical to achieving defined ecosystem objectives. For instance a flood may need to last for a longer period, rather than increasing the flood peak itself. Thus, environmental flows will almost certainly be different from a certain minimum or average flow. Variability in the natural flow regime of a river comprises five key components—magnitude, frequency, duration, timing, and rate of change. These are recognized as being essential for sustaining an ecosystem's biodiversity and integrity (Poff and Ward 1989, Rosenberg et al. 2000).

Variability is necessary to sustain geomorphological dynamics, as well as longitudinal and lateral connectivity. A certain degree of flexibility by setting different flows at different times of the year has been instigated in many instances, but there is also a need to incorporate year to year variations such as wet and dry years. However, in dry years, it may be necessary to give priority to power interests at the expense of environmental objectives.

Environmental flows should be seen in relation to other potential and current remedial measures. For instance, modifications to river channel morphology and substrate characteristics, the location of in-stream structures such as weir and groynes, the construction of fish ladders and bypass channels, as well as changes in dam construction, such as multilevel dam off-takes, can make important contributions toward improving ecosystem function (Brittain and Nilsson 1996).

Most modern hydropower schemes incorporate at least some degree of environmental flows. However, there are many old schemes put into operation at a time when environmental considerations were certainly not to the fore and in many cases not considered. Nevertheless, many of the schemes are required to renew their licences after a number of years of operations. This may vary from 20 to 100 years. Licence renewal in many cases provides the opportunity to include environmental flows into project operation. Another opportunity may be in connection with maintenance and upgrading of dams to meet new safety requirements. The economics of upgrading may indeed be so substantial that decommissioning is an option. However, simply removing a dam does not necessarily bring the river back to its original state. There will have been geomorphological changes as a result of the regulated hydrological regime. In addition floral and faunal communities will undoubtedly have changed. Some of these changes may be irreversible. There will also be the problem of accumulated sediments in the reservoir. These will either have to be physically removed from the reservoir or flushed downstream in a managed manner to avoid damage to downstream biota and to other user interests, such as water supply (WCD 2000, IUCN 2003).

Methods and solutions

The setting of environmental flows has been the subject of considerable interest internationally and several countries are addressing the problem. There has been a gradual change from the lack of any compensation flows at all, prevalent in many of the pioneering hydropower schemes in the early 1900s. Then in the 1950s and 1960s, minimum flows below large dams were instigated, followed by in-stream flows in the 1970s, hydrological and habitat based methods in the 1980s to today's multidisciplinary catchment-based criteria used to develop environmental flows. However, in many countries the notion of more or less fixed and constant minimum flows is still prevalent and there is a clear need to think more in terms of environmental flows.

There is a huge wealth of methods and approaches to setting environmental flows, extensively reviewed by several authors in recent years (e.g., Stewardson and Gippel 1997, Dunbar et al. 1998, Arthington and Zalucki 1998, Arthington et al. 1998, Tharme 2003). However, as requirements and context vary enormously, there is no single best method, approach, or framework to determine environmental flows (IUCN 2003). Methods for the setting of environmental flows can, however, be allocated to a number of main categories with increasing degree of complexity and resource requirements (Dunbar et al. 1998, Tharme 2003).

a. *Hydrological methodologies,* also termed "look up techniques" are the simplest and are frequently based on one or more hydrological indices, such as specific proportion of average discharge. Hydrology-based methods sometimes include catchment variables (e.g. O'Shea 1995) or are modified to incorporate on the basis of hydraulic, biological, and geomorphological criteria (e.g., Estes 1996 and Tennant 1976, respectively). These techniques are widely used and require a relatively low level of resources. Such methods are undoubtedly of value in an initial screening process or in low conflict situations.

b. *Hydraulic rating methodologies* utilize a quantifiable relationship between the quality of an in-stream resource, such as fishery habitat, and discharge, to calculate flows (e.g., Stalnaker and Arnette 1976). These examined the effects of specific increments in discharge on in-stream habitat, with most emphasis placed on the passage, spawning, incubation, rearing, and other flow-related maintenance requirements of individual, economically important fish species. Hydraulic rating methodologies use changes in simple hydraulic variables, such as wetted perimeter or maximum depth, as a surrogate for habitat factors known or assumed to be limiting to target biota. The implicit assumption is that ensuring some threshold value of the selected hydraulic parameter will maintain the biota or ecosystem integrity, or both.

c. *Habitat rating or simulation methodologies* attempt to assess environmental flows on the basis of detailed analyses of the quantity and suitability of in-stream physical habitat available to target species or assemblages under different flow regimes, making use of integrated, hydrological, hydraulic, and biological response data (Petts and Maddock 1994). Typically, the flow-related changes in physical microhabitat are modeled in various hydraulic programs, usually using depth, velocity, substrate composition, and cover, collected at multiple cross-sections within the river study reach. The simulated available habitat conditions are linked with information on the range of suitable to unsuitable microhabitat conditions for target species, life stages, assemblages, and activities, often depicted using habitat suitability index curves. The final outputs, usually in the form of habitat-discharge curves, and habitat time and duration series for the biota, are used to predict optimum discharges. These techniques, also termed biological response modeling, are usually the most resource intensive, but they are considered more defensible, although not without their problems (Gore and Mead 2000). Within this category, the In-stream Flow Incremental Methodology (IFIM) is the most widespread. One of the elements in IFIM is PHABSIM (Physical Habitat Simulation). These techniques are primarily based on physical variables, but are linked to the physical requirements of fish. Considerable efforts are being made to improve these techniques and to include invertebrates (e.g., Brunke et al. 2001).

d. *Discussion based approaches and hydrological analysis*. There has been an increasing tendency to use expert opinion in the setting of environmental flows in combination with hydrological time series comparing historical, natural, and alternative flow regimes. Such an approach also includes holistic methods that have been particularly well developed in Australia and South Africa, where the whole river system, including the river channel, riparian zone, and groundwater, is the focus for field assessment, hydrological modeling, and multidisciplinary workshops (Arthington 1998, Arthington and Zalucki 1998). In certain cases, instead of starting with no discharge and then determine what is necessary for specific uses, one starts with the maximum acceptable deviation from the norm, i.e., how much water can one remove without producing significant geomorphological or ecological damage.

A Case study—the Norwegian salmon river, Suldalslågen

The Norwegian salmon river, Suldalslågen, has been exploited for hydropower since 1966. Much of the water flowing out of the large lake, Suldalsvatnet, is diverted through power plants directly to the fjord. The river, about 20 km long, is well known for its large Atlantic salmon (*Salmo salar*) and the size and timing of the spring flood has been shown to influence both smolt migration and survival of fry emerging from the gravel. Before regulation, the spring flood regularly reached 400 $m^3 s^{-1}$ and even exceeded 700 $m^3 s^{-1}$ in some years. After regulation, the magnitude of the spring flood has been substantially reduced. The river has been regulated in two stages. The first (1966–79) resulted in higher discharge in winter and lower in summer, while the second regulation (1982-91) reduced flows in winter and spring.

In order to understand the effect of the changed flow regime on the salmon population, two different flow regimes are being instigated, 3 years with a moderately high discharge (1998–2000) during spring and 3 years with a low maximum discharge during spring and an autumn flushing flow (2001–2003). Normally, in regulated rivers the timing of the spring flood is fixed, while in this particular trial experiment in Suldalslågen, it was adjusted to natural conditions during the first period. Using a reference site in an unregulated adjacent catchment, the spring flood was realized at the same time as high flows in the tributaries of Suldalslågen. This meant that the timing of the spring flood differed by over a month in 1998 and 1999.

A reduction in the spring flood did increase water temperatures and led to increased growth in young-of-year (Y-O-Y) fish, although there was no increase in winter survival due to larger size. However, the smoltification age was reduced, which gave higher smolt production. It is possible that the main limiting factor for fish survival is lack of preferred habitat, especially during winter.

In such rivers one of the major problems after regulation is the change in sediment transport. Reduced flows cause fine material to accumulate and fill the interstitial spaces, creating poor habitat for juvenile fish. Aquatic mosses have also increased substantially in Suldalslågen because of the absence of major floods; another factor that was envisaged would lead to habitat deterioration. Thus, in the second trial period, an autumn flushing flood was introduced. Preliminary results suggest that an autumn flood does not produce significant changes in substrate conditions compared to "normal" regulation regime with a spring spate. The autumn flood removed moss from summer fish habitat, but it was replaced by sand. However, the response time may be long in such full scale experiments. The generation time of the biota differs widely from one or more generations per year in many aquatic insects to a generation length in the order of 3-8 years in salmon. Because of the inherent uncertainties in predicting complex hydrological and biological systems, the trial period should be long enough to see the full effects of the proposed environmental flow regime.

Gaps in our Knowledge and Constraints to Progress

The constraints to progress in the field of environmental flows and hydropower lie within two main areas, management and science. Management is often conservative and, even though new ideas and concepts are accepted, formal changes in guidelines and legislation may take many years to implement. Nevertheless, increasing pressures both from the public at large, from scientists, and from bodies such as the European Union are forcing changes in the management of rivers.

There is an increasing move toward catchment-based management plans and environmental river flows are an essential element in any such plan. Stakeholder involvement and public awareness also come into play in such a situation and there are often clear financial constraints to modifying river flows in regulated rivers. Nevertheless, it may be possible to modify the discharge regime in several ways without significant reduction in the hydropower potential. For example, it may be possible to release more water in wet years and less in dry years, or adjust the timing of floods. Technical installations may also be limiting to the implementation of environmental flows. For instance, old dams may only have one release gate, thus fixing the nature of the outflow water. In certain regulated rivers, such as those with large-sized substrates, such as rocks and boulders, unrealistically high flows may be needed before aesthetical goals can be met.

From a scientific perspective, lack of ecological knowledge is a major stumbling block to the implementation of environmental flows and as long as knowledge of the aquatic environment remains limited, setting threshold environmental flows will inevitably retain an element of expert judgment. Very often, we simply do not know what critical flows determine ecosystem function and integrity. In many cases expert judgment is the nearest we can get to an objective and quantitative determination of environmental flows. There are frequently problems in extrapolating from one catchment to another, each river being unique in its characteristics. This is not only a problem within biology, but the determination of hydrological characteristics from ungauged catchments is still being explored. We do have quantitative methods for a number of key organisms, such as salmon and trout, at least at the reach scale. However, for many ecosystem components, we lack the knowledge to determine thresholds and evaluate which characteristics of the flow curve are critical. Can we assume that what is good for target species such as salmon is good for other ecosystem components? Clearly different species and even different life stages will have varying requirements and even competing demands as regards flow. The requirements for salmon fry will be different from adult salmon and different again from, for instance, waterfall vegetation. Another aspect is that many of the species dependent on high flows, such as plants growing in the vicinity of waterfalls, are of high conservation value and are often on national and international Red Lists.

There has been an increasing pressure to develop small-scale schemes. However, the requirements for assessment of environmental impacts are usually much less than larger schemes, on the assumption that that the impact on the environment is significantly less. This may be true, but little research has been done either on the effect of individual projects or on the synergic effects of several projects on the same watercourse. It may be that larger schemes, with their increasing attention to environmental flows and other mitigations, are in fact less of an impact on the river environment than numerous small schemes.

5 On Flow Variability and Stream Ecosystem Evolution

Introduction

Many studies support the idea that flow variability is a major driving force in shaping fluvial hydrosystems. Flow variability is included in several conceptual templates that predict species richness, paying attention to their ecological strategies (Southwood 1977, Hildrew and Townsend 1987, Townsend 1989). They are all based on gradients of resource utilization versus limitation in a framework of disturbance characteristics and levels of productivity (Stazner and Higler 1986). In general, a variable and unpredictable flow regime will stress biotic functions and, conversely, a stable flow regime will favor biotic interactions. Hence, a pragmatic objective has been the determination of a required or vital in-stream flow regime that will support, or allow recovery of, a river's functional processes. Present methods, however, reveal our fuzzy knowledge of the flow variability effect on stream ecosystem dynamics because they are based on "trial and error" implementation of water management plans (Richter et al. 1997).

Human activity interacts with stream ecosystem dynamics over a range of scales. However, large-scale impacts concern mainly breaks in the continuity of the stream energy flow with well identified and localized sources of impairment. For mid-size basins of hundreds of square kilometres, where human activity impacts faintly or sensitively but continuously on the hydrological fluxes, a more meso-scale process-based approach is required to identify the key hydrological processes that could sustain, enhance, or limit ecosystem evolution. To achieve this goal, functional processes have to be related to different geomorphic and hydrological contexts (Poff 1997), and related to the disturbance regime that has to consider the implications for river food webs as well as the hydraulic processes for sediment transport. There are some large-scale stream ecological studies (Statzner and Higler 1986, Lamouroux et al. 2002) that confirm the filter effect (Poff 1997) of local geomorphic characteristics (reach slope, geomorphic unit succession, grain size distribution, bank hydraulic conductivity) on benthic species distributions. Frequent flow variability triggers the input and output of energy fluxes within the water column and the hyporheic domain (Evans and

Petts 1997) and infrequent (but not rare, see Breil 1997) floods provide key ecological pulses (Junk et al. 1989). Frequent flow variability is like an engine that induces and enhances exchanges between the water column and the bottom sediment. Frequency, intensity, duration, and direction of the fluxes depend on the sequence of geomorphic features as well as hydrological fluctuations.

Ecological Dimensions of the Flow Regime

Several ecological strategies or ecological traits have been proposed to explain the adaptation of stream ecosystems to flow variability. In summary, there are (a) competing species, adapted to very specific conditions in a stable environment, (b) colonizing species adapted to unstable environments, and (c) organisms that have the ability to colonize severe and predictable environments. The habitat template concept (Southwood 1977) predicts that species richness peaks in the center of a template based on the change in flow versus flow unpredictability. Predictable and small changes in flows correspond to a spring stream type, while highly unpredictable and small changes in flows characterize headwaters. Predictable and high changes in flows is a third category that includes desert streams and pro-glacial flow rivers. Predictability, or timing, is an important feature of the flow regime because specific flow conditions may be required for sensitive live stages, such as spawning and emergence, that occur at given times. In the Intermediate Disturbance Hypothesis (Connell 1978, Ward and Stanford 1983), axes are replaced by intensity and frequency of environmental disturbances (not specific to flow). The peak richness is expected at an intermediate distance level on each gradient because only specific strategies are adapted to the extremes. In the disturbance–productivity–diversity model (Hildrew and Townsend 1987), both productivity and disturbance of stream conditions are expected to modify the benthic community species richness. Richness is low for low levels of disturbance and productivity, moderate with high productivity and low disturbance, and high with high disturbance and productivity levels. The Patch Dynamics Concept (Townsend 1989) is a template designed by both the spatial and temporal dimensions, where competition, predation, and colonization occur under the regulation of frequency of disturbance leading to a “cellular” and “competitive lottery” model. This conceptual template seems to be flexible enough in space (then in time) to adapt to a meso-scale approach and is supported by the implicit notion of disturbance.

Disturbance

The last two models introduce the frequency of disturbance that can be quantified from a discharge time series if we can define clearly what are the disturbance processes. Disturbance theory is a result of the observed persistence or stability of ecosystems over time in different places with different environments (Connell and Wayne 1983). Two viewpoints are discussed: (a) the necessity to invoke an equilibrium between an ecosystem’s stable state with its variable environment and (b) the buffered capacity of an ecosystem against extinction. The latter can be supported only by biotic compensation based on an existing species pool. In that case, species density variations over time are second order factors for species pool persistence. Stability then results from a multiple stable states, each being an expression of the same persistence. The main question then arises as to the appropriate scales of time and space that are needed to judge ecosystem stability. Too fine a scale will always exhibit unstable states in response to discrete, punctuated abiotic disturbances, and larger scales will always tend to stability by averaging information (Connell and Wayne 1983). The minimum time scale is the turnover time of an assemblage (expected to represent a functional trait). For the spatial scale, the minimum area is that which provides all required conditions to implement a complete live cycle. Disturbance definition can only be considered by keeping these scales in mind if we want to reconcile the persistence of a sustainable river ecosystem with human water use.

Controversial arguments have been deployed to clarify the definition of disturbance predictability or unpredictability for a stream ecosystem (Resh et al. 1988, Poff 1992). Frequency of disturbance is assumed to constrain in time the stream ecosystem dynamic equilibrium. From an ecological point of view, a disturbance is assumed to be a “Destructive, rapid or prolonged change in the physical environment, which exceeds (a) the normal range of conditions experienced by a substantial number of organisms in a population or community or (b) the rate of their ability to adjust, resulting in their death and/or removal.” A major disruption can occur in the ecosystem life cycle as a response to a major change in the geomorphic environment (Cattanéo et al. 2001). Infrequent but not rare flows can be assumed to be necessary and positive disturbances for stream ecosystems. For example, the bank-full discharge that can vary in frequency among streams from several times a year to several years (Poff 1992) shapes geomorphic features and rejuvenates mesohabitats. In sandy bed rivers, surface benthos is often absent but deep hyporheos often exhibits great activity because of high rates of exchange between underground and free surface running waters through the sandy matrix (Boulton 1993, Rouch et al. 1997, Fellows et al. 2001). From these examples, one can say that disturbance definition depends on both species traits and abiotic features.

The consequences of a regulated flood regime on the stream ecosystem dynamic are not a matter of one species because linked food chains can lead to compensatory effects with, for example, a decrease in growth rate at the same time as a decrease in competitors, or as a result of a change in the feeding strategy of end-chain predator species. A predictable disturbance regime would mean the ecosystem having adapted to, or being continuously constrained by, this event in its overall life cycle. An unpredictable disturbance is assumed to occur anytime in the ecosystem cycle, leading to an unstable equilibrium with perhaps multiple stable states. One can expect that a predictable disturbance regime will offer more opportunity for a functional assemblage to develop than an unpredictable disturbance regime but this is only an assumption. Does it mean, anyway, that only natural regimes, with their specific disturbance regime, are able to maintain some basic processes that sustain a stream ecosystem in a healthy state (Townsend and Riley 1999)?

Do we need a near-natural flow variability?

Considerable ecological research supports the premise that healthy aquatic and riparian ecosystems depend upon maintaining some semblance of natural flow regime (Petts 1996, Richter et al. 1997). A main challenge for the future concerns the quantification of temporal disturbance that is presented as a common regulating factor in ecological theories (Minshall 1988). There is now a desire to use the overall flow records that exist for managing rivers and streams as an ecological resource, and not just as a water resource (Clausen and Biggs 2000, Petts 1996, Richter et al. 1996). To do this, several authors (Clausen and Biggs 2000, Poff and Ward 1989, Poff 1992, Poff and Allan 1995, Breil 1997) have proposed sets of flow variability characteristics to group hydrological regimes from an ecological perspective. These characteristics integrate both normal and extreme conditions: (a) normal conditions are described using mean flow, median flow, skewness of the frequency curve distribution, coefficient of variation and predictability indices (Colwell 1974), and (b) extremes are described in terms of the upper 10 and 20 percent quartiles, extreme flow frequencies, mean durations, total duration and volume over given discharge threshold, ratio of the extreme flow magnitudes to the median flow, and predictability indices. A main objective is to define from these groups the biotic processes sustaining the ecological integrity in streams (Petts 2000). It would then be possible to define the required flow conditions to maintain the timing, intensity, and duration of the basic processes that depend on the water and linked material fluxes. Another key structuring factor is water temperature, which governs the timing of important biotic functions. Temperature exhibits a predictable annual pattern that should be considered in any assessment of flow variability types (Harris et al. 2000, Petts 2000), not least in relation to climate change scenarios.

Discussion

Spatial and temporal flow variabilities are linked at a broad scale. However, expected patterns have not always been confirmed. Some studies (e.g., Statzner and Higler 1986) have focused on species richness, but this is a poor indicator of trophic resource utilization. Production and respiration would be the most appropriate indicators, but are often unavailable. The inadequacy of biotic descriptors seems to be a major reason why researchers have failed to demonstrate ecological theories for species richness prediction, because they rely on an often unverified spatial representation (Minshall 1988). Then the role of landscape filters, from the broad to the geomorphic-unit scales, provides an heuristic framework to understand the distribution and abundance of species in streams (Poff 1997). Filters are biotic and trophic limiting factors whose expression can differ from along the scales. In this framework, it is assumed that a significant density of a species belonging to a regional pool can only occur if that species possesses appropriate functional attributes (species traits) that allow it to accommodate all the upper scale filters from regional to local. Following this framework, we have to identify which flow characteristics “control” which ecological characteristics from the regional scale, dominated by a climatic regime, to the mid-size basin scale which is dominated by sub-regional geological characteristics, and then to the reach scale, which is mostly dependent on the local gradient. As an example, fish species structure can be very sensitive along the up- to down-stream gradient for smooth relief regions. At this scale, flow variability can exhibit a large range of values, from very large to very limited for basins with areas in the range of 500 to 1000 km^2 (unpublished data for France). However, flow variability often exhibits spatial patterns that reveal the dominant role of the geology, which controls, in turn, the relief characteristics. This is the scale where human pressures can be easily identified and corrected if required. This is also the scale where integrated water management can take place.

To mimic the natural flow regime is the best way to maintain basic ecological processes. However, given the artificial influences on contemporary river systems, the priority is to identify the required timing, frequency, magnitude, and duration in flow variability that will sustain a coherent succession of processes in space and time. Important processes in the transition zones between the water column, the banks, and the bottom must not be overlooked. A stable flow regime will inhibit these exchanges, imposing a one-way transfer of water, nutrients, and energy. Bioenergetic and hydraulic research studies on these ecotones are necessary, not least to advance our knowledge of the role of flow variability on the budget, accumulation, and transformation of carbon and nitrogen in streams (Dent et al. 2001).

Conclusion

To complete and enhance our knowledge of flow variability in streams, three advances are required:

a. Regional ecohydrological analyses to define climatic controlling effects on flow regimes in mid-sized basins. Human influences would not be sufficient to modify the regional climatic pattern that supports a regional pool of species, but mid-sized basins would exhibit different sensitivities to human pressures.

b. In the mid-sized basins, land uses and flow variability must be appropriate indicators of ecological status using the principle of fuzzy logic and table scores.

c. Refined research studies must be developed to focus on local processes that are repeated all along a river, propagated by sequences of geomorphic units, when exposed to the same flow variability.

6 Observations on Environmental Flows in Headwater Streams: The Girnock Burn, Scotland

Introduction

There is currently considerable interest amongst river managers in re-naturalizing flow regimes in regulated rivers to sustain or enhance ecological status. A range of hydrological indices have been recently suggested as metrics to describe natural flow regime variability (Richter et al. 1996b, Poff et al. 1997), characterize flow regimes (Harris et al. 2000), or assess changes resulting from land use change (Archer and Newson 2002). However, at a practical level, there remain major issues as to which methods or metrics to use (Olden and Poff 2003). Moreover, such methods are rarely tested against ecological data (exceptions being Claussen and Biggs 1997, 2000); they focus exclusively on water quantity rather than water quality, and they assume that gauging sites provide representative perspectives on the upstream catchment.

In this note, various published and unpublished works on the Girnock burn in northeast Scotland are used as a basis for exploring the ecological significance of flow variability in an upland environment, where there are long-term ecological data sets to correspond to hydrological and water quality data sets in a relatively undisturbed catchment. Limitations in the existing hydrological assessment methods are highlighted and a call is made for a more comprehensive, holistic approach to understanding the influence of hydrology on freshwater ecosystems.

The Girnock Burn

The Girnock burn drains a small, 30-km^2 subcatchment of the river Dee in northeast Scotland (Langan et al. 1997). The catchment spans an altitudinal range from 230–570 m, and is underlain by a complex suite of granite and metamorphic geologies. Peaty soils predominate and support heather (Calluna) moorland with small areas of forestry. Mean annual precipitation is 900 mm, with 550 mm of runoff, leaving evapotranspiration losses at 350 mm. The Girnock burn is an important spawning burn for Atlantic salmon and fish populations have been monitored since 1966.

Hydrology

The flow regime of the Girnock burn is remarkably flashy (Soulsby et al., in prep.). Flood peaks can exceed 50 cumecs and return to base flows very rapidly. The annual flow regime has been classified according to hydrograph shape and magnitude using the approach of Harris et al. (2000). Annual flow peaks occur in December–January, February, or March and years can be characterized by high, intermediate, or low flows. However, there is no correspondence between the timing of peak flows and the overall flow magnitude, probably because the Scottish Cairngorms occupy a climatically transitional (subarctic) zone with very variable weather conditions.

An analysis of the frequency, timing, and magnitude of individual hydrograph peaks showed no correspondence to the analysis of annual hydrological regime (Soulsby et al., in prep). Thus, flow indices that have been found to be ecologically significant in the Scottish Cairngorms, such as the frequency of discharges 3 and 7 times the median flow (Gibbins et al. 2001), did not correspond to "wet" years. In other words, a different perspective on ecologically important hydrological conditions was gained at the annual and event scale.

Ecological response to flows

Uniquely in the Girnock, high resolution ecological data are available to analyze against hydrological parameters. Two particular life stages of Atlantic salmon that are flow-sensitive are those of spawning (Moir et al. 1998) and smolting (Youngson 1983). PHABSIM modeling at a number of control reaches in the burn have shown how variable flow conditions during the spawning season (typically 25 October to 22 November) can affect the spatial and temporal availability of spawning habitat (Moir 1999). In some years spawning habitat is widely available throughout the catchment for most of the spawning period (Webb et al. 2001, Gibbins et al. 2002). At other times, spawning might be restricted to one or two hydrological events and is thus spatially and temporally constrained (Moir et al., in prep.). In particular, high flows are needed to allow fish to access the upper parts of the river system and provide suitable spawning conditions (Moir et al., in prep.). This is because different reach types have differing sensitivity to flow changes as far as spawning suitability is concerned.

Despite the consequent effect on the number of spawning fish and egg deposition, implications for smolt production are limited. Annual smolt production from the Girnock is remarkably constant (2000–2500 fish), though the timing of smolting may exhibit strong hydrological cues during the spring and autumn (Gibbins et al. 2002). Although strong hydrological cues influence both spawning and smolting, the standard flow indices suggested in the literature (e.g., Olden and Poff 2003) are insufficient to capture these, as narrow biologically mediated time-windows do not correspond to the time periods (e.g., monthly) used in standard hydrological analysis.

Importance of water quality and environmental change

The thermal regime of rivers is, in many ways, as important as the flow regime in influencing ecological processes (Gibbins et al. 2002; Hannah et al., in press). Analysis of the temperature regime (in terms of timing and magnitude (Harris et al. 2000) in the Girnock revealed no relationship to the hydrological regime (Soulsby et al., in prep.). Moreover, analysis of long-term temperature data (post-1968) from the Girnock showed that winter and spring temperatures have increased by ca. 1°C over the period of record. There is some evidence that these changes are influencing the timing of smolting (i.e., that occurs earlier in the year, with the main concentration of spring smolts occurring in April rather than May). Moreover, the modal age of smolts is now 2 years, rather than 3 years, suggesting the warmer weather results in more rapid growth, allowing the physiological changes associated with smoltification to occur earlier. Whilst these interactions are still being elucidated, temperature demonstrates that water quality needs to be considered as well as water quantity, if the ecological significance of annual regimes in physico-chemical parameters is to be understood.

In addition to the physical parameter of temperature, chemical water quality parameters are an important aspect of hydrological variability in upland streams (Soulsby et al. 1998). This is particularly apparent in the Girnock, where the spatial variability in geology results in variability in stream chemistry (Malcolm et al. 2004). In some granite-dominated parts of the catchment, acidic water conditions prevail at high flows and such conditions are less suitable for salmonids than other parts of the catchment, where calcareous rocks give rise to well-buffered stream waters. In addition, in some areas, groundwater upwelling results in deoxygenated conditions in spawning gravels and poor egg survival (Malcolm et al. 2004). This shows that hydrological influences on stream ecology are often very subtle at a range of spatial and temporal scales. Thus, it is perhaps not surprising that hydrometrically based flow regime descriptors are relative insensitive to ecological responses.

Conclusions

While it would be churlish to claim that the hydrological indices methods produced in the literature are not a useful step toward helping to renaturalize flow regimes in heavily regulated river systems, there is a clear need for such methods to be tested against ecological data—both at seasonal and event scales. In addition there is a need to recognize the importance of critical, biologically important time “windows,” which may not be detected in classification and variability indices. In addition the spatial texture of channel types dictates that fluvial geomorphology results in differential impact on flow variability in different sectors of a catchment, which may be assessed by a single downstream gauging station. Furthermore, in upland environments, hydrological variability may be associated with water quality variability, which may have ecological significance. Again, this has generally not been considered by published assessment methodologies. Finally, as temperature regimes in the Girnock show, nonstationarity in flow and hydrochemical regimes need to be recognized in any ecological assessment.

7 Observations on the Ecological Functioning of Temporary Headwater Streams and Springs: The English Peak District

Introduction

Headwater streams and springs have been poorly studied by lotic ecologists, despite their high frequency of occurrence, potential contribution to species richness and diversity (Feminella 1996, Hoffsten and Malmqvist 2000), and high level of endemism in some locations (Erman and Erman 1995). The role of flow permanence (hydroperiodicity) on the ecology of intermittent headwater streams and springs has been widely recognized, although its potential overriding influence on biotic community structure is still poorly appreciated (Smith and Wood 2002). Headwater streams and springs demonstrate the majority of the structural and functional properties seen in other lotic systems, yet are considered significantly less complex than sites further downstream (Williams and Williams 1998). They represent ideal locations to examine the relationships between biotic communities and the environmental parameters that influence their distribution.

A wide variety of springs and headwater streams exist, ranging from those draining largely impervious upland catchments to lowland groundwater-fed systems. Most headwaters and springs support floral and faunal communities that are distinct from those further downstream (Smith 2000). However, our current state of knowledge regarding headwater and spring ecosystems is limited in several respects: (a) a large volume of published records represent data from a few (less than five) spring and stream sites (e.g., Boulton and Lake 1992, Hayford and Herrmann 1998) and the examination of multiple sites at a regional scale is rare (e.g., Lindegaard et al. 1998, Erman 1998); (b) temporal variability in community composition at individual sites is largely unknown (Gooch and Glazier 1991); and (c) few data are available for intermittent or ephemeral sites, and, although specialist temporary water taxa are recognized from many locations (Boulton 1989, Williams 1987), their ecology is poorly understood. The management of headwater streams and springs poses a number of problems owing to the paucity of data and a history of regulation for multiple uses, including mineral water supply, irrigation of crops and livestock, and, in some instances, as ornamental landscape features.

This paper illustrates the influence of perennial and intermittent flows on the macroinvertebrate communities of (a) limestone (karst) springs and (b) intermittent and perennial springs and headwater sites in the English Peak District. Groundwater fed springs and streams provide a unique interface between surface and subterranean habitats, supporting populations of both epigean and hypogean taxa (Botosaneanu 1998, Sket 1999) and potentially constituting refugia for relict fauna (Williams and Williams 1998).

Headwater Springs of the English Peak District

Invertebrate and habitat data relating to three different flow levels were obtained from a total of 48 springs (34 perennial and 14 ephemeral) draining limestone within the English Peak District, subsequently referred to as the White Peak, and on headwater sites on the River Lathkill (Wood et al., in prep.).

The springs in the White Peak had broadly similar water chemistry and displayed relatively little variability in the physical parameters examined. Water temperature was within 1.2°C (range 7.8–9.0°C) of the mean annual air temperature in the area (8.0°C), with the exception of one thermal spring with a mean water temperature of 18°C. Preliminary analysis indicated that no significant differences were recorded between the perennial and intermittent springs at high discharge. Macroinvertebrate community composition was highly variable within perennial springs at low flow, depending of the total volume of flow decline at individual sites (ranging from 26 to 85 percent), although all intermittent sites were dry at this time. Examination of the community data collected at intermediate flow, when six highly ephemeral springs were dry, provided the clearest discrimination between invertebrate communities from perennial and intermittent sites. These data were used for subsequent analysis of variance.

Detrended Correspondence Analysis (DCA) indicated that there was a core of macroinvertebrate taxa present in springs across the White Peak and that the intermittent sites were located on the periphery of the ordination (Figure 3). One-way analysis of variance demonstrated that there was a significant difference between the number of taxa, log-transformed community abundance, Shannon-Wiener diversity index, and the Berger-Parker dominance index for perennial and intermittent springs at intermediate discharge (Table 2). For the 42 springs examined at intermediate discharge, the number of taxa (Figure 4a) and the Shannon-Wiener diversity index (Figure 4b) were lower at intermittent sites than perennial sites. However, log-abundance (Figure 4c) and the Berger-Parker dominance index (Figure 4d) were significantly higher for intermittent springs than perennial springs.

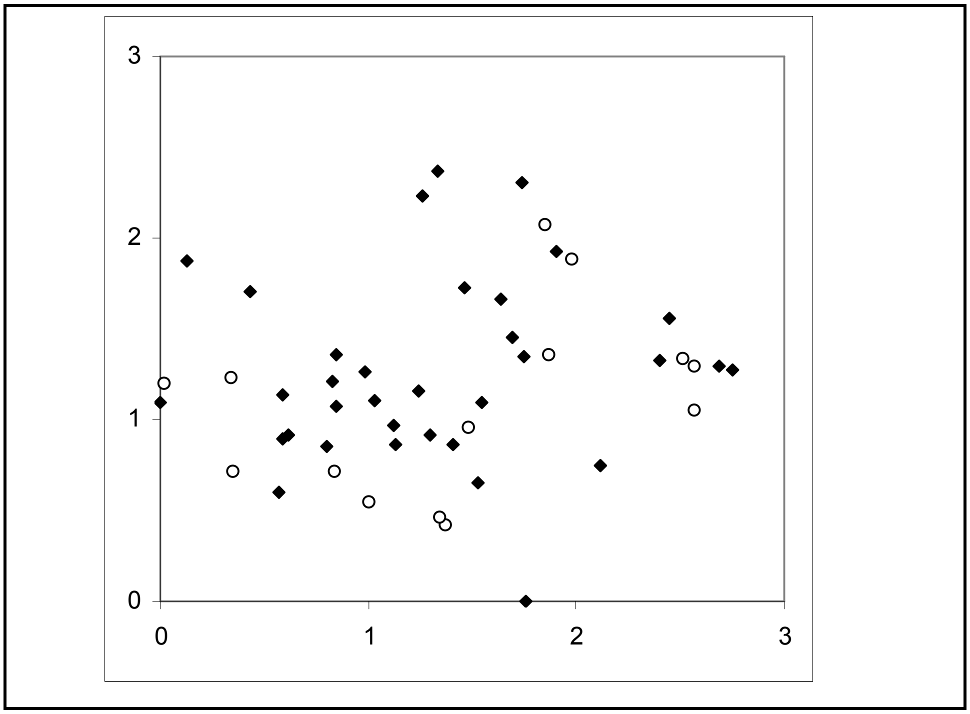

Figure 3. Detrended correspondence analysis (DCA) site biplot of 48 springs in the White Peak (1998–1999). Solid symbols = perennial springs; open symbols = intermittent springs

Table 2
One-Way Analysis of Variance of Ecological Indices from Perennial (*n* = 34) and Intermittent Springs (*n* = 8) in the White Peak (total *n* = 42) at intermediate Discharge (1999)

	Mean square	***df***	***F*-ratio**
Number of taxa	53.54	1	4.55*
Shannon-Wiener	3.88	1	19.46***
Log-abundance	6.69	1	4.53*
Berger-Parker	0.67	1	16.07***

* P<0.05; *** P<0.001

DCA of samples from springs and headwater sites on the River Lathkill indicated that intermittent springs and perennial headwater sites formed relatively distinct groups with limited overlap (Figure 5). However, perennial springs and intermittent river sites formed a mixed overlapping cluster at the center of the ordination. One-way analysis of variance indicated a significant difference between the number of taxa and log-abundance of invertebrate communities from intermittent and perennial springs and main-stem sites at intermediate discharge (Table 3). However, no differences were recorded for the Shannon-Wiener diversity and Berger-Parker dominance index. The number of taxa and log-community abundance was lowest at intermittent sites and highest at perennial river sites (Figure 6).

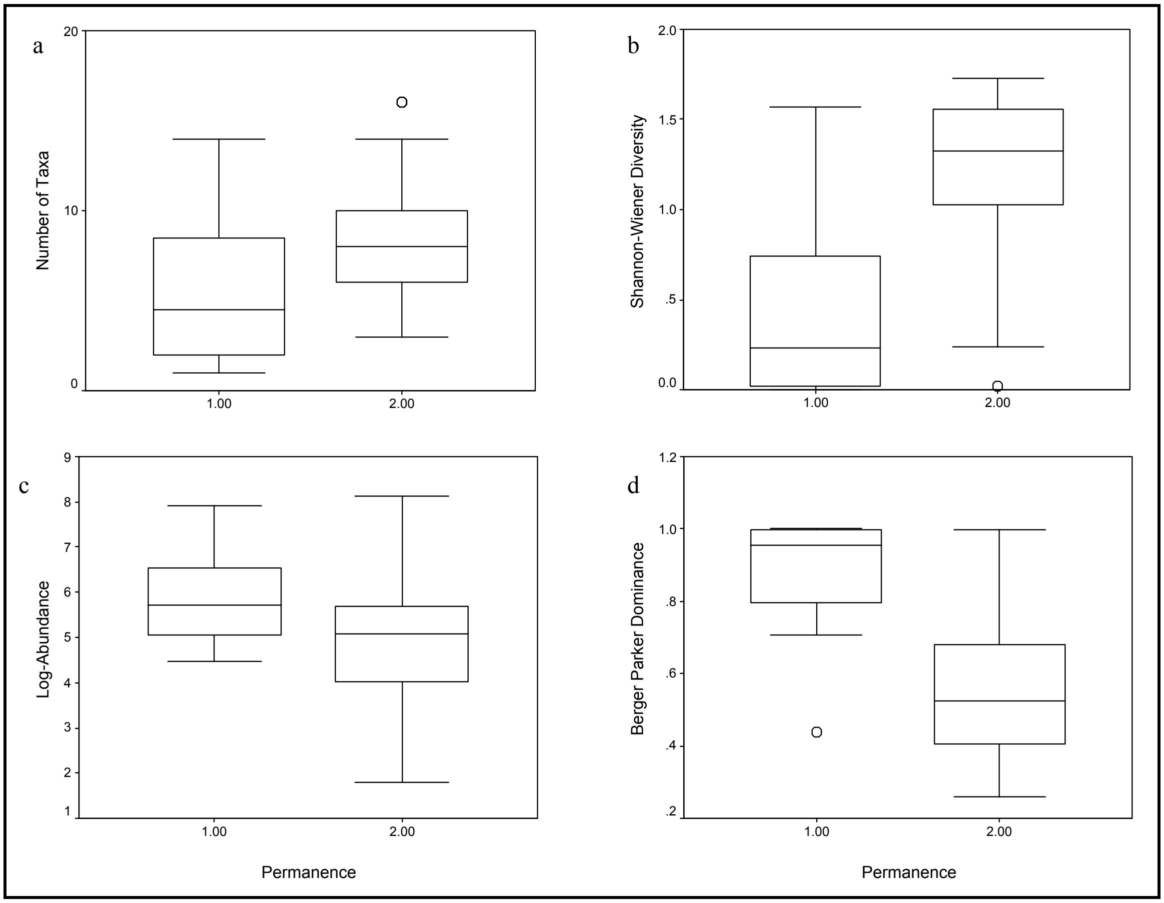

Figure 4. Box-plots of invertebrate community indices for intermittent and perennial springs from the White Peak at intermediate discharge: a) number of taxa; b) Shannon-Wiener diversity index; c) Log-community abundance; and d)Berger-Parker dominance index. 1= intermittent springs; 2 = perennial springs; O = outlier

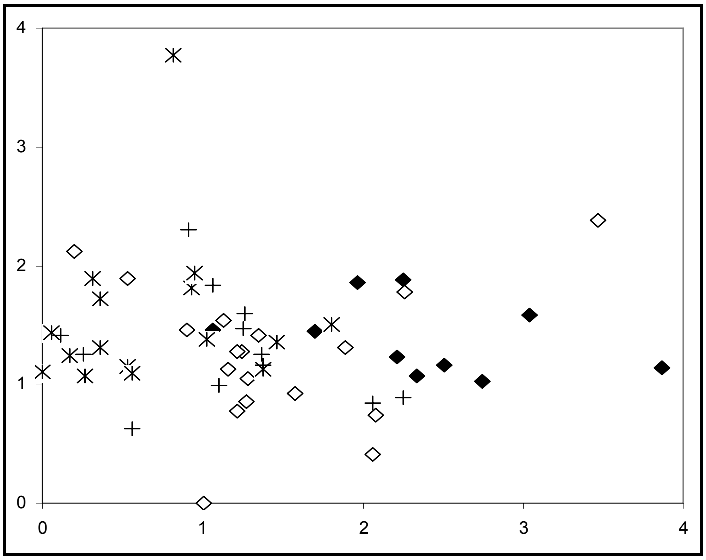

Figure 5. Detrended correspondence analysis (DCA) site biplot of intermittent and perennial springs and headwater sites in the River Lathkill catchment (1998-2000)

Table 3
One-way analysis of variance of ecological indices from intermittent springs (*n* = 10), perennial springs (*n* = 18), intermittent river (*n* =12); and perennial river (*n* =16) within the River Lathkill catchment (total *n* =56) (1999–2000)

	Mean square	**df**	***F*-ratio**
Number of taxa	73.63	3	6.14**
Shannon-Wiener	0.01	3	0.05
Log-abundance	15.68	3	6.72**
Berger Parker	0.01	3	0.35

** P<0.005

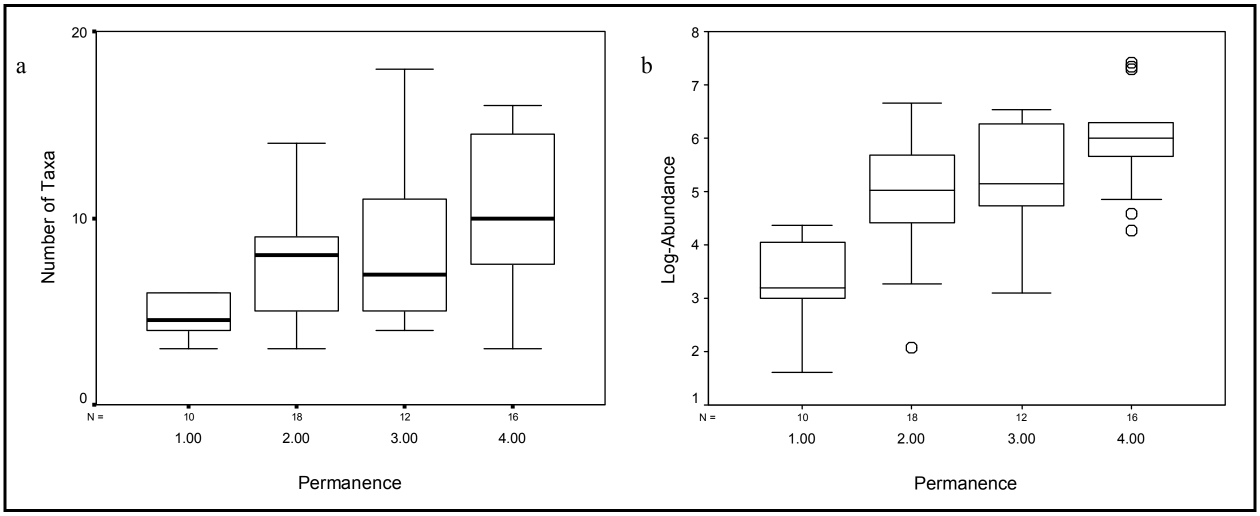

Figure 6. Box-plots of invertebrate community indices for intermittent and perennial springs and headwater sites in the River Lathkill catchment at intermediate discharge: a) number of taxa; and b) Log-community abundance. 1= intermittent springs; 2 = perennial springs; 3 = intermittent river; 4 = perennial river; O = outliers

Discussion

It is widely assumed that intermittent headwater sites and spring systems support reduced numbers of taxa, and support lower diversities and abundances of invertebrates compared to perennial sites and those further downstream (McCabe 1998, Smith and Wood 2002). Detailed analysis of the aquatic invertebrate communities in the 48 springs across the White Peak using DCA identified a core of relatively common and ubiquitous fauna found in a wide range of aquatic habitats including springs, small streams, and rivers (e.g., *Gammarus pulex*; Amphipoda: Gammaridae) and taxa able to utilize intermittent sites (e.g., *Stenophylax permistus*; Trichoptera: Limnephilidae).

The relatively low number of aquatic taxa and low diversity recorded at intermittent sites at intermediate discharges almost certainly reflects the ability of individual taxa to colonize and sustain populations in intermittent aquatic habitats (del Rosario and Resh 2000). Insect taxa able to aerially colonize springs, exploiting resources accumulated during the dry phase, have a competitive advantage over noninsect groups and aquatic taxa unable to withstand periods of flow cessation (Covitch et al. 2003). These rapid colonizers, particularly Chironomidae, were very abundant at intermittent sites with the resumption of flow in the springs and explain the high Berger-Parker dominance index values within these sites. In contrast, perennial sites were characterized by a greater abundance of the Amphipod, *Gammarus pulex*. However, *G. pulex* was not excluded from all intermittent sites, reflecting its ability to rapidly recolonize sites from sources downstream and the presence of subterranean populations 'upstream' within the well-developed groundwater drainage network in parts of the White Peak (Gunn et al. 2000).

The invertebrate communities recorded from intermittent and perennial springs and headwater sites on the River Lathkill also reflect the ability of taxa to colonize and sustain populations at sites with intermittent flow. DCA suggested an environmental gradient that reflected flow permanence but also location within the stream network. While there was some degree of overlap between perennial and intermittent springs and mainstream sites, the perennial springs and intermittent mainstream sites in particular formed an overlapping group at the center of the site biplot. This reflects the location of the intermittent mainstream sites in relation to the perennial river and, more importantly, to perennial springs, some of which form tributaries of the intermittent headwater river and allow rapid recolonization of taxa from upstream sources with the resumption of flow.

The number of taxa recorded within the perennial and intermittent springs of the River Lathkill reflects the broader pattern recorded over the White Peak at intermediate discharge; although there was not a significant difference between perennial springs and intermittent mainstream sites. In marked contrast to the 42 springs from the White Peak, invertebrate abundance recorded in the intermittent springs in the Lathkill catchment was significantly lower than for perennial sites at intermediate discharge. This is a result of the large area of the Lathkill catchment that experiences intermittent flow and as a result a reduction in the colonization potential of aquatic taxa owing to the increased distance between source water bodies. This also emphasizes the importance of knowledge regarding the hydrology and ecology of the wider drainage network.

The management of springs and headwater sites has been largely neglected and is fraught with potential problems. Structural management of in-stream and riparian habitats can have significant impacts on the resident communities and may eliminate, or significantly degrade, any conservation interest (Smith and Wood 2002). A greater understanding of the physical resource is required, incorporating knowledge of the wider catchment characteristics, land use, and hydroperiodicity of individual sites. This will almost certainly enable a more informed knowledge of the biological resources recorded at individual sites and help in the understanding of differences among sites. The invertebrate communities of intermittent headwater sites and springs are probably at greatest threat from inappropriate management operations. Careful consideration needs to be given to the aquatic flora and fauna present, but also the semiaquatic and terrestrial fauna that utilize these habitats as flow declines and ultimately ceases. These latter groups of taxa have not been considered in this study or most historical studies of springs and headwater streams. The inclusion of these taxa, particularly those known to utilize semiaquatic habitats, such as some Diptera and Coleoptera groups (Drake 2001, Lott 2001), is essential in future studies. This will almost certainly increase the number of taxa and diversity recorded at these sites, and will ultimately change the widely held misconception that intermittent springs and headwater river sites have low scientific and conservation value because of biological impoverishment.

8 A Commentary on River Productivity

Introduction

Knowledge of freshwater food web structure is a prerequisite to understanding and managing fluvial hydrosystems. A traditional view of river systems holds that fish and their invertebrate prey rely on inputs of nutrients and carbon from the surrounding catchment. This is probably the case in many small forested streams where shading by riparian trees limits in-stream primary production. Much less is known about food webs in larger rivers, although a number of conceptual models have been advanced that make predictions about their structure and function.

Perspectives on Food Webs

Three models place contrasting emphasis on the importance of autochthonous versus allochthonous carbon for river food webs. The first of these, the river continuum concept (RCC) (Vannote et al. 1980), is well known and has made a significant contribution to our understanding of running freshwaters. In particular, it emphasizes the importance of terrestrial organic matter derived from upstream processing for the trophic economy of downstream river reaches, and downplays the role of in-stream primary production, asserting that algal growth is limited by high turbidity and light attenuation. In contrast to the RCC, the flood-pulse concept (FPC) (Junk et al. 1989) stresses the importance of lateral river–floodplain exchanges for the food web. Essentially, the FPC postulates that the aquatic metazoa migrate onto floodplains to exploit terrestrial resources and then return to the main channel when flood waters subsequently recede. In contrast to the RCC, the FPC emphasizes the importance of floodplain inputs to the river food web. It is undoubtedly the case that the physical linkage between rivers and their floodplains is a fundamental factor determining the productivity of many pristine river systems. Nonetheless, many naturalised rivers have become divorced from their floodplains by aggressive management, and are thus denied the terrestrial inputs yielded by the flood-pulse cycle. The third and most recent concept, the riverine productivity model (RPM) (Thorp and Delong 1994), was originally intended to depict food web processes in these highly impacted, large river systems. The RPM flatly confronts the predictions of both the RCC and the FPC because it places great emphasis, not on allochthonous organic matter from the catchment, but instead on in-stream primary production as the basis of the riverine food web.

The RPM

Thorp and Delong (1994) contended that "the primary, annual energy source supporting overall metazoan production and species diversity in mid- to higher-trophic levels of most rivers (>4th order) is autochthonous primary production entering the food web via the algal-grazer and decomposer pathways." The model was initially formulated to represent the functioning of highly regulated rivers with limited floodplains like the Ohio River. The importance of in-stream primary production to the food web of the Ohio River was confirmed using stable isotope techniques (Thorp et al. 1998). Most recently, Thorp and Delong (2002) revised their RPM model, extending its predictions to include unregulated, floodplain rivers.

The RPM is challenged by the observation that riverine respiration frequently exceeds net primary production. How can animal biomass in large rivers be fuelled primarily by in-stream primary production if the ecosystem as a whole is heterotrophic? Thorp and Delong (2002) argue that this "heterotrophy paradox" can be resolved for river systems where respiration exceeds production *if the food web functions as two essentially independent, or weakly linked, pathways* (Figure 7). The authors argue that in excess of 90 percent of total organic matter is never ingested by metazoans but metabolised within the microbial loop, whereas relatively small quantities of labile algal carbon are ingested by herbivores, and subsequently channelled up the herbivore–algal pathway to predators. It is slowly becoming accepted that the majority of allochthonous carbon *is* processed within the microbial loop (Sinsabaugh and Findlay 2003), but the importance of benthic algae and phytoplankton to the metazoan food chain remains controversial.

A number of recent studies based on stable isotope analysis have tested the importance of algal carbon to the metazoa of large rivers. In a 15 year study of the Orinoco River food web, Lewis et al. (2001) found that phytoplankton and periphyton were the major carbon source, even though 98 percent of available carbon was from other sources (CPOM or macrophytes). Similarly, Bunn et al. (2003) confirmed that algal carbon supported the food web of an arid zone floodplain river in Australia. However, not all food web analyses support the RPM; for example, Angradi (1994) found that algal production was important to fish in only one of three Colorado River tributaries.

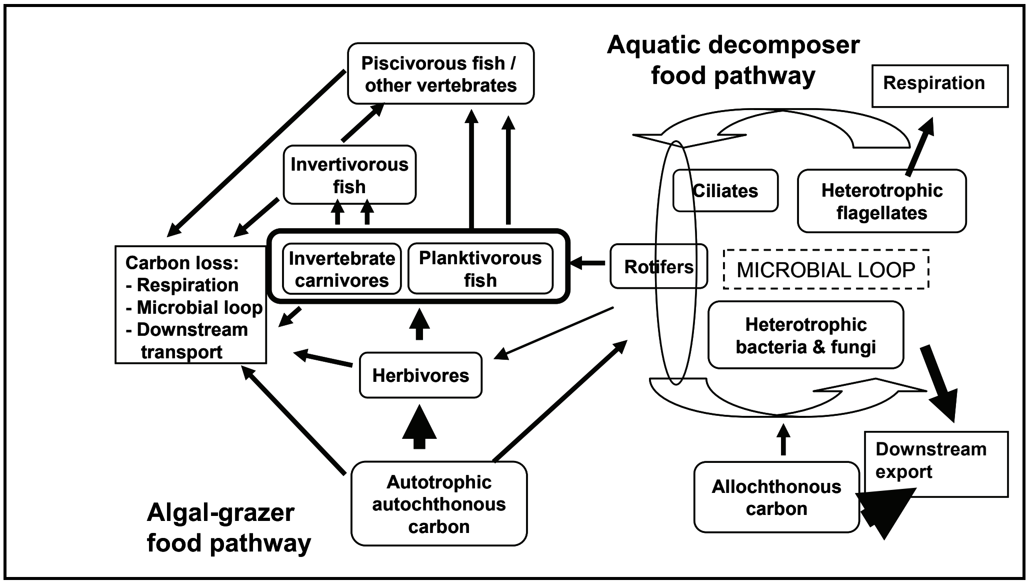

Figure 7. Aquatic food pathways (after Thorp and Delong 2002)

An interesting and controversial facet of the RPM is that it downplays the importance of detritivorous metazoans, particularly invertebrate shredders. In headwater streams, abundant shredders comminute detrital particles, making them available to collector–gatherers and filter feeders downstream. Importantly, they also channel detrital carbon to invertebrate predators and fish. The omission of the detrital carbon pathway from the RPM implies that large river food webs operate as two distinct sub-webs or compartments: the algal–grazer–predator sub-web and the detritus–bacteria–microinvertebrate sub-web, or microbial loop. Other sub-webs associated with extensive microhabitat types may also exist in large rivers. For example, stands of macrophytes support species of Trichoptera, Orthocladiinae, and Simulidae not present on mineral surfaces, and this may also be the case for communities that are largely water-borne as opposed to those essentially benthic in habit. Clearly, the spatial extent of large rivers increases the likelihood that compartments exist in their food webs, and this structure may have consequences for the stability of the system.

A Way Ahead

The RPM makes a number of assumptions about food web structure in large rivers that contravene our view of freshwater food webs in general and imply that web structure changes significantly from headwaters to mouth. In addition to a high degree of compartmentalization, we may infer from the RPM that, in large river food webs, specialized herbivorous feeders dominate the primary consumers while detritivorous species are scarce. Thus, in toto, decreased omnivory, reduced connectance, increased web height, and more skewed linkage strength may be features of large river webs not replicated in fishless headwaters, but which confer decreased stability to the lower reaches of the drainage network. Ecologists have invested much effort in resolving food webs in small streams. This is entirely justified, but that research effort must now be replicated downstream in large river systems that may have very different dynamics to the upper reaches. Our focus must now fall on (a) the trophic base of production in large rivers, (b) the distribution and strength of trophic links in space, (c) the vulnerability of tall, narrow sub-webs to trophic cascades, and (d) the mysteries of the microbial loop. Finally, all three models mentioned in this note take a bottom-up approach that ignores population dynamics entirely. We should advocate a combined approach to food web science that illuminates patterns of energy flow and predator–prey interactions in the future.

9 A New Look at Dissolved Organic Matter

Introduction

Until recently, dissolved organic matter (DOM) was seen just as a residue of biological activity, similar in concentration and composition in all environments, and therefore inert. However, as technology has improved, our ability to characterize DOM in the environment has increased, and observations have been made that suggest that DOM is actually more labile and more variable. A new paradigm has emerged, where DOM is a more interactive component of aquatic ecosystems, and where similarities in DOM do occur across ecosystems, reflecting a myriad of biogeochemical processes rather than inertness. Of course, many issues remain. For example, how "inert" and "reactive" is different DOM from different sources? What is more important to an ecosystem—a large pool of relatively inert DOM, or a small pool of reactive DOM? How does human influence affect either?

DOM in the aquatic system depends on a wide range of factors that are shown in Figure 8. Allochthonous DOM input (quality and quantity) depends on landscape, vegetation, hydrology, and climate. This DOM is then cycled, at a wide variety of temporal and spatial scales, as part of the "bacterial loop." Autochthonous DOM may also be generated within the aquatic system; the relative proportion is poorly understood and will also vary with vegetation, hydrology, and climate. DOM will be utilized by the bacterial community in a manner that will vary with the nature of that community, both in terms of its physiology, ecology, and phylogeny. Finally, some or most of the DOM will be exported from the system—usually this is from a river to the ocean DOM pool. A rapidly growing literature exists that attempts to better understand the complex interactions that occur in Figure 8 at a wide range of temporal and spatial scales: the reader is referred to Findlay and Sinsabaugh (2003) for greater detail. This short note attempts to summarize the state-of-play in terms of the current state of knowledge and practice, examples of good practice, and constraints to progress.

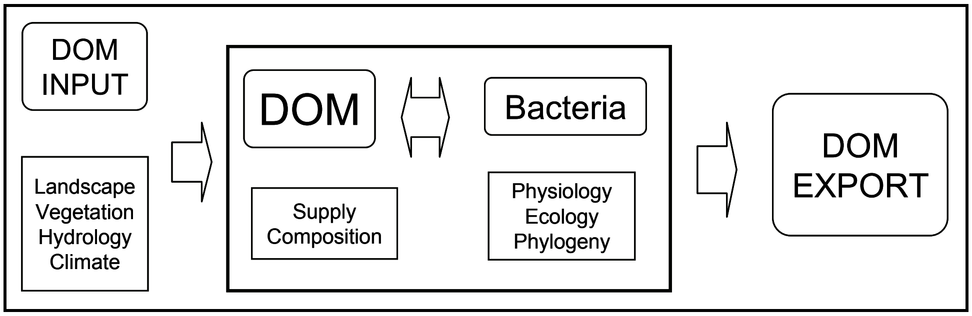

Figure 8. DOM in aquatic systems. (After Sinsabaugh and Findlay 2003)

DOM: Current State of Knowledge and Practice

The current knowledge and practice in determining DOM can be divided between that of DOM quality and DOM quantity. Quantity is easily measured by proxy, by determining either total organic carbon (TOC) or the absorbance of water at a set wavelength (typically around 400 nm), or both. Studies of global rivers have shown some consistent spatial patterns; for example, higher concentrations of DOM are found in catchments with a high proportion of peat soil type. Very recently, interest in the United Kingdom has focused on temporal variations in TOC and absorbance. In river catchments that drain upland peat lands in the United Kingdom, a 65-100 percent increase in water color has been observed to have occurred over the last 20 years but the cause of this change is unclear. Does the color increase reflect a DOM concentration increase or quality change? Is the cause higher summer temperatures or land use change or both? What are the implications for ecosystem function, or carbon budgeting, for example?

In contrast to DOM quantity, DOM quality (for example molecular weight or chemical structure) is harder to measure. Many biogeochemical techniques are invasive and require pre-concentration. This is time consuming, expensive, and can alter the structure of the DOM under investigation. For example, Kaiser et al. (2003) use solid state and multidimensional solution-state nuclear magnetic resonance spectroscopy to identify chemical groups and chains within the DOM. However, these analyses required the concentration of 100 L of river water with a DOM extraction of only 12–48 percent. Therefore, although the determination of DOM chemistry greatly assists in our understanding of DOM function, one is never sure the extent to which it is the same chemistry that occurs in the natural environment. Unfortunately, in-situ techniques (such as the measurement of absorbance) provide little “quality” data. Rapid developments in the measurement of DOM fluorescence do have significant potential in this area (Baker 2001, 2002). DOM can be analyzed in less than 1 minute, less than 0.5 mL of water is needed, detection limits are in the ppb to ppm range (depending on DOM type), and the technique is noninvasive and potentially automatable (as demonstrated by Rainer et al. 2003).

As well as requiring further knowledge of DOM quality and quantity, as introduced earlier, the source (allochthonous vs. autochthonous) and bioavailability (labile vs. recalcitrant) of DOM are also not well understood and require further research.

DOM: Key Principles and Good Practice

For routine analysis, DOM is actually measured by "proxy," using established techniques that are used to set water quality targets and to detect organic pollution. Although not technologically state-of-the-art, these techniques provide reproducible data over long time periods. However, they have limited use for understanding DOM quality, quantity, source, and bioavailability

Quantity can be determined using total organic carbon analyzers, as discussed earlier, or by absorbance. Color, as measured using absorbance, correlates with concentration and so is only an indicator of DOM quantity, event though its measurement is often driven by "colored water" being seen as a DOM quality issue. A wide variety of measurement techniques (Hazen color comparators, absorbance measured at a variety of wavelengths) over time makes time series data difficult to interpret.

Quality is hardly determined using existing good practice. For example, biochemical oxygen demand (BOD) is the closest to providing data on DOM bioavailability. However, it is time consuming, costly in staff time, and low in reproducibility. Ammonia, nitrate, and phosphate are also often measured as water quality determinants and can also provide limited data on DOM quality data and source. Other more advanced biogeochemical techniques used include high performance liquid chromatography (HPLC) and gas chromatography–mass spectroscopy (GC-MS). These are now being used to detect trace DOM pollutants (pesticides, endocrine disruptors, pharmaceuticals, PAHs, etc.). However, for many, detection limits are close to environmental concentrations and again the techniques are time consuming and expensive.

DOM: Gaps in Knowledge and Constraints to Progress

The previous sections give a flavor to the gaps in knowledge and the constraints to progress in terms of our understanding of dissolved organic matter. Five areas are highlighted here:

a. *DOM structure.* Is there a model DOM structure? Is there a model molecular weight range for DOM? How does DOM structure vary with landscape, land use, vegetation type, and ecosystem? Is there a difference between headwater DOM (which should all be autochthonous) and downstream DOM (which is likely to have a greater allochthonous component)? How does autochthonous and allochthonous differ (if at all) in terms of structure?

b. *DOM–ecosystem interaction.* Does DOM quality and quantity determine the bacterial community? Or does the bacterial community determine DOM quality and quantity? When and where does the former happen? When and where does the latter happen?

c. *DOM bioavailability.* What is more important to ecosystems: small amounts of labile DOM or large quantities of recalcitrant DOM? Does DOM lability change though time and space, and if so, why and how?

d. *DOM–land use and landscape variations.* How does land use affect DOM quality and quantity? Is the location of the land use (landscape metrics) important with respect to the aquatic system?

e. *DOM export.* Marine scientists calculate that only 1 percent of all terrestrial DOM reaches the sea. Is there a missing carbon store in the oceans? If not, where does the terrestrial carbon go?

10 Modeling Fish Population Dynamics and In-Stream Flows

Introduction

During the last 20 years, a number of studies have dealt with the relationship between fish (especially salmonids) and physical habitat availability (Bayley 2002), often in relation to the definition of in-stream flow downstream from dams. Their common goal has been to demonstrate the biological significance of 'carrying capacity' (estimated with habitat models such as PHABSIM, Bovee 1982) as a limiting factor of population size. However, is it possible to translate a carrying capacity value into population status without understanding the linkages between habitat and biological processes (e.g., reproduction, energetics, mortality)? Of course not, because all environmental factors vary when measured in the field; "the challenge is to understand how this variability affects population dynamics" (Rose and Cowan 1993), i.e., to demonstrate relations of cause and effect among fluctuating processes (population demography, habitat suitability for different life stages during life cycle) or with a quantitatively fuzzy variable (carrying capacity of streams). Thus, recent studies have assessed and quantified the effect of habitat variability on population dynamics (Cattanéo et al. 2002). According to Railsback and Harvey (2003), modeling fish populations could be useful (a) to assess effects of flow regimes, not just static minimum flows, (b) to assess cumulative effects of changes in flow, temperature, and other factors, and (c) to produce population responses to alternative flow regimes (directly applicable to decision-making).

Thirty years ago, the importance of natural stream flow variability (magnitude, frequency, duration) in maintaining healthy aquatic ecosystems (biodiversity) was virtually ignored in a management context (Poff et al. 1997). Now, the objective of population dynamic models integrating cause and effect in fish habitat relationships is to define flow management strategies satisfying stream and human water needs (Cardwell et al. 1996, Whittaker and Shelby 2000).

This paper is an overview of how population dynamics has become indispensable simultaneously in research and management contexts.

Current State of Knowledge/Practice

Fish habitat relationships

Numerous studies have dealt with environmental limiting factors that could affect fish populations in relation to stream physical habitat suitability (Fausch et al. 1988). Fausch et al. (1988) studied 99 models predicting fish biomass versus abiotic (and especially habitat) variables. They demonstrated the lack of transferability of models because of too many parameters and too few samples, and certainly because such approaches were too static to describe dynamic processes.

The IFIM concept aimed to find a solution to this lack of temporal aspect in habitat description by including discharge variability. Even though the model has been the subject of considerable controversy in the literature, PHABSIM (in IFIM, Bovee 1982) is nonetheless widely used (Tharme 2003). It is also one method that enables coupling hydraulic data and biological data (such as habitat preference) to simulate the evolution in the potentially available habitat (known as Weighted Usable Area (WUA)) for an aquatic species, as a function of discharge. However, despite its widespread use throughout the world, PHABSIM has not yet been well validated in biological terms (Lamouroux et al. 1999; Kondolf et al. 2000). The different reasons for this have been well rehearsed (Pouilly and Souchon 1995). One of the reasons is the difficulty in obtaining reliable hydrological and biological time series covering a sufficiently long period of time to study population responses to habitat temporal variability, attempting, for example, to identify thresholds of magnitude, duration, and frequency defining habitat limiting periods or carrying capacity (Capra et al. 1995). Then authors presented simulation models that place the changes in habitat into a population response context, which was a more promising alternative according to Williams (1999).

More recently, predicting fish population response to flow variability (and suitable habitat availability) has been the main objective of some studies (e.g., Studley et al. 1996 (the Altered Flow Project)), and some of them demonstrated cause and effect relationships. The main results suggested that stream discharge in winter (Cattanéo et al. 2002, Lobon-Cervia 2003), or during the second half of winter (Mitro et al. 2003), is the major determinant of annual recruitment in trout populations. This result, observed from a large number of streams, could be considered as the main habitat influence on population dynamics because recruitment explained the main part of the spatio-temporal variation in cohort size (Rose and Cowan 1993, Lobon-Cervia 2003).

However, such results could be biased owing to the lack of population demography analyses. Indeed, effects of disturbances on fish habitat and populations depend on the pre-disturbance condition of in-stream and riparian habitat, timing of the disturbance, and life histories of individual species (Dolloff et al. 1994). For example, the assumption that enhanced minimum in-stream flow for fisheries should result in the production of more or larger fish was not always supported (Harris et al. 1991). In situations where factors other than minimum low flow are limiting populations, enhanced fish production is unlikely to be observed. A minimum flow regime developed to protect spawning habitat and egg incubation may limit densities of older age classes (observed 3 years after dam construction by Scruton and Ledrew 1997). On the contrary, even if high flows favor adult abundance, all age and size classes may not benefit from the higher flows.

Modeling fish population dynamics

Even if physical habitat is generally assumed to be the most important stream limiting factor, population dynamics modeling was at first based on demographic and density-dependent regulations only, such as stock-recruitment models (Elliott 1994, Williams 1999). But Elliott (1994) concluded his book dealing with 30 years of research on brown trout ecology (essentially based on population dynamics), in specifying that “definition of habitat requirements, seen in terms of a multidimensional niche, is a high priority for future research on brown trout.” This conclusion was a warning for future research on population dynamics modeling that must integrate physical habitat as a main variable in the same way as population demography.

Two other main approaches exist to simulate population dynamics.

Population dynamics is considered as the average behavior of all individuals within a population. These models translate a conceptual model of population dynamics into a numerical form. Some deterministic models which were used to study fish population demography were developed for the whole population, such as stock-recruitment models (Elliott 1994, Williams 1999). Other deterministic models are based on the Leslie matrix (a population is divided into groups of equivalent individuals: e.g., age classes). Deterministic model development is often associated with complete sensitivity analysis of all model parameters over a long period of time and with analysis of balanced population structure.

The environmental influence was progressively introduced into such models from the 1970s. Recently, they have taken into account the role of spatial fragmentation (Charles et al. 1998), the role of habitat availability (Gouraud et al. 2001, Hilderbrand 2003) or the ecotoxicological impact (Chaumot et al. 1999) on population dynamics. The geographical distribution of individuals was taken into account when migrations of individuals between different patches can influence the global demographic process (Charles et al. 1998). For example, these authors used the discrete case of the ‘variable aggregation method’ (with aggregated matrix; Sanchez et al. 1995) for which the main assumption is that migration time scale (day) should be much faster than the demographic one (year). Hilderbrand (2003) studied the role of carrying capacity on resident cutthroat trout populations in a fragmented habitat using a very simple age-structured matrix model. He showed with a sensitivity analysis that the more restricted a population’s capacity to expand is, the greater the extinction risk. Modypop (Sabaton et al. 1997; Gouraud et al. 2001), a matrix model based on age classes, simulates change in a trout population (the numbers in each age class calculated over time) at a stream scale, using biological parameters that are dependent on environmental conditions. A 1-month time step is chosen so as to take into account climatic variations and their impact on the population. Among the central hypotheses of the model, two biological mechanisms have been selected by which fish populations are structured, based on the variability of limiting factors, as suggested by Maïki-Petäys et al. (1999). The first relates to limitation of the adult trout biomass by the amount of habitat available during summer low-water periods (Gouraud et al. 1999, 2001). The second possible limitation relates to a decrease in the numbers of young of the year due to high discharge between their emergence and their first summer (Latterell et al. 1998), recently confirmed by Cattanéo et al. (2002) on a large scale (30 stream reaches in France). The demographic parameters (survival, fertility, growth rates, displacement) are dependent on temperature, trophic availability in the environment, and carrying capacity (Bovee 1982).

The limiting factors in developing deterministic models are that:

a. They usually needed advanced knowledge in mathematics and in programming, which is not necessarily within the reach of every biologist.

b. They do not enable users to predict, or to compare their results, to individual behavior (response).

Population dynamics is considered as the sum of the behavior of each individual within a population. The individual-based approach is "an explanation of the systems properties by referring to the properties of single individuals" (Kaiser 1979) and their interactions. Individual Based Model (IBM) is a recent concept that bloomed during the 1990s (Van Winkle et al. 1993, Ginot et al. 2002) based on individual animals, which took advantage of the increase in computer power (object programming). This is what Grimm (1999) called the "pragmatic motivation," because it seems like a new tool in the toolbox of ecological modeling. According to IBM philosophy "the population and community-level consequences would emerge naturally" from individual properties (DeAngelis et al. 1994), which was called paradigmatic motivation by Grimm (1999).

IBMs are also spatially explicit (mobility), using either continuous or discrete space. IBMs consider that individuals within a population are distributed in space and that important interactions among individuals take place over some predefined local scale. In cellular automata models (category of IBM), the change in the state of any one cell from one time step to another depends on its own state and the state of some number of its neighboring cells (Molofsky 1994). Generally, the time step could vary from 1 day to 1 week, or more, depending on the question being addressed (Van Winkle et al. 1993).

Different types of IBM exist :

a. Multiagent systems (autonomous object, controlling its own behavior; Ginot et al. 2002).

b. Mechanistic models (numerous parameters and equations; Jager et al. 1997, Van Winkle et al. 1998).

c. Bioenergetic models (Hayes et al. 2000, Hughes 2000, Essington 2003) modeling fish movement (and swimming speed), growth, food intake (drift foraging models), metabolic rate.

d. The approach of Williamson et al. (1993) and Bartholow et al. (1993), which resulted in SALMOD software, was a 'middle ground' between IBM and aggregated classical population models.

Numerous questions have already been addressed with IBMs. Jager and her colleagues worked, for example on:

a. Trout persistence and climate changes (1999).

b. Factors controlling white sturgeon recruitment and test of extinction risk (2001).

c. Designing optimal flow for chinook salmon (2001, 2003).

The limiting factors in developing IBMs are:

a. The conceptualization of the natural processes and the potential role of differences among individuals.

b. The availability of appropriate and statistically valid data upon which to formulate the rules to put into the models (causes and effects).

c. Their complexity, which can be assessed by counting the full number of parameters a model uses (often more than 20 parameters among papers presented in Grimm's review [1999]).

Key Scientific Principles for River Managers and Examples of Good Practice in Applying These Principles

Much research has been done, and more is needed, to produce a rigorous understanding of the complex relationship between flow, fish population, fish catch, and economic value (Harpman et al. 1993). Castleberry et al. (1996) argued that no scientifically defensible method exists for defining the in-stream flows needed to protect particular species of fish or aquatic ecosystems. They recommended an approach of adaptative management. Establishing in-stream flows involves scientists (to develop monitoring methods for adaptative management, based on a more secure biological knowledge) and resource managers (to accept the existing uncertainty regarding in-stream flow needs) with challenging roles in the process (Castleberry et al. 1996). But the important variability of population dynamics among streams suggests that greater understanding of production ecology is required before stream salmonids can be managed on a sound basis (Lobon-Cervia 2003). The following studies dealing with population dynamics and flow management demonstrate clearly the relevance of new management tools.

Example 1

While there are numerous examples of aquatic population models in the literature, few of these models explicitly capture the effect of stream flow and lend themselves to the analysis of different flow management regimes. The study by Jager and Rose (2003) is an example of prediction under alternative flow management regime. They studied the life history diversity of pacific salmon (metapopulation structure), which decreased due to habitat degradation. The model they used simulated optimal flow regimes (in m^3 yr^{-1}) versus natural (and diversion) flow and management objectives (seasonal flows pattern that maximize recruitment or spawning time variation). They used a numerical optimization technique coupled with a recruitment model (IBM: ORCM; time step = day) to design optimal seasonal flow patterns. Their results showed that regulating flows in a manner that would conserve a wider range of run times would produce fewer total recruits than would regulating flows in a manner that maximizes total recruitment. Finally, they underlined that the optimal flow did not necessarily mimic the natural flow pattern and that the role of flow pattern could be less important than the elimination of barriers to migration, for example.

Example 2

IFIM is now widely used in France in impact studies on hydropower installations, to determine the in-stream flow to be recommended for the bypassed sections to meet the requirements of water resource legislation as far as salmonids are concerned. The French National Guaranteed Flow Working Group (representatives from EDF, government agencies, research bodies) has worked since 1994 on predicting fish population response to in-stream flows changes. As for Studley et al. (1996), this study is based on ecological (populations, habitat, discharge) monitoring before and after in-stream flow change. But the French project integrated the development of a population dynamics model to understand population responses to an increase of habitat availability (defined for example by WUA) expected after an increase of in-stream flow. Experiments are now under way on eight bypassed sections. For one of those eight sites, Capra et al. (2003) showed the importance of using a population dynamics model to understand population response to natural or artificial (bypassed) flow regime. In the population dynamics model (Gouraud et al. 2001) habitat was integrated on several levels (mean habitat conditions for potential breeders in low-flow periods, mortality when the population in the area exceeds the available carrying capacity, mortality among fry when discharge is high in the post-emergence period (Cattanéo et al. 2002). Capra et al. (2003) showed that high flows after emergence were the main environmental factor limiting the number of the young-of-the-year in the bypassed section as in the natural stream part. Habitat availability before or after increase of minimum flow is not responsible for population structure change.

These two examples show that population response to different flow regimes could only be studied with a dynamic approach, integrating simultaneously habitat (discharge) time series and population demography. That is, for population dynamics models, integrating environmental variability is certainly the best solution to define a long-term flow management strategy, if the influence of environmental variability is represented by relations of cause and effect (e.g., high flows limiting 0+ density).

Future Management

Research shows that varied flow regimes provide a diversity of aquatic species and life stages, create and maintain a diversity of channel features, and provide a diversity of ‘niches’ for recreational activities (Whittaker and Shelby 2000). Today, scientists and agencies focus on protecting flows for several resources, and the integration of flow needs for multiple values (Whittaker and Shelby 2000). But this requires an increase in model complexity. Could this alternative be a reasonable management tool? In the future, models will have also to reflect the effects of temperature (climate change) on fish populations (e.g., Daufresne et al. 2003). Recent models already integrate population responses to water quality variability (Chaumot et al. 1999, Scheibe and Richmond 2002). Such models are essentially useful for developing and testing concepts, for future research trends. However, as long as they cannot be validated with field observations, their use as tools for management decisions cannot be recommended.

Surprisingly, at the same time, different authors discuss the significance of preference curves for different discharges (Ibbotson and Dunbar 2002, Armstrong et al. 2002). Even if such an assumption is important for the significance of models that attempt to predict the effects of stream flow change on fish habitat (Shirvell 1994), population responses are of greatest interest in defining future management strategies. Then it could be interesting to test the sensitivity of population dynamics models to habitat suitability model outputs to progress this debate.

Models of multispecific community dynamics could be easily developed now, because some biological validations of the influence of habitat availability on fish community structure exist (Lamouroux et al. 2002, Daufresne et al. 2003). Fish population dynamics models could be transferred to fish community dynamics models. Moreover, the recent development of a simplified habitat description methodology (Lamouroux and Capra 2002, Lamouroux and Souchon 2002) will be very useful to work at a larger scale than the stream reach, being more compatible with population or community long-term studies.

The topic “population dynamics modeling and in-stream flow” is still in its infancy, as it is too soon to assess the long-term effects of proposed alternative in-stream flow solutions based on population responses. Thus, Jager and Rose (2003) specified that their “results have not been verified by empirical studies.” Moreover, there is now a strong tendency to reduce the mean number of years spanned by studies (Bayley 2002). In this way, it will become very difficult for a researcher working individually to find relations between cause (environmental changes) and effect (population responses). Researchers will have to work together to facilitate data and modeling experience exchanges. As was clearly demonstrated by Cattanéo et al. (2002), quantification of relationships between habitat and population structure could be obtained only with long-term multisite sampling.

11 The Ecological Functioning of Exposed Riverine Sediments

Introduction

An important challenge in hydroecology is to develop effective management strategies for riparian environments (Naiman and Decamps 1997). However, such an aim requires a fuller understanding of the interrelationships among the natural processes that drive riparian disturbance regimes.

The riparian habitat is strongly influenced both by channel dynamics and the frequency of flood events (Ward et al. 1999, Tockner et al. 2000), and this (often seasonal) disturbance is thought to maximize biological processes (Naiman and Decamps 1997) and both in-stream and riparian biodiversity (Naiman et al. 1993, Naiman and Decamps 1997, Ward 1998). There is a wealth of research on the important structuring role of disturbance on aquatic communities of rivers systems (e.g., Lake 2000). However, the role of predictability and stochasticity in natural disturbance processes impacting upon terrestrial communities in river corridors has not received the level of attention that it warrants (Plachter and Reich 1998). In part, this reflects the fact that natural river corridors are home to a complex range of aquatic and terrestrial organisms (Robinson et al. 2002), which makes generalizations extremely difficult. This is particularly the case with terrestrial invertebrates on exposed riverine sediments (ERS) (gravel or sand bars, and shoals) that are associated typically with highly dynamic rivers with unregulated flow regimes.

ERS environments are predominantly inorganic, have little vegetation, and are characterized by high fluxes of temperature and humidity across the bare sediments. Thus, ERS are not only hostile environments for most mobile terrestrial organisms but they appear to be maintained by secondary production (Hering and Plachter 1997, Hering 1998). Although ecologists have begun to identify the faunal communities found within these highly disturbed habitats (Reich 1991, Eyre et al. 2001b, Sadler et al. 2004), little research exists that links invertebrate dynamics (community and population) directly to the variation in physical habitat dynamics.

This paper will: (a) argue that ERS environments are unique in riparian ecology, as they exhibit both high productivity and high species richness coupled with high levels of species fidelity and rarity (Sadler et al. 2004), and (b) review the current status of the knowledge of ERS ecology, suggesting areas that require further study and evaluation.

Current Status of ERS Science

The potential importance of naturally active river corridors as ecosystems that support high levels of gamma (landscape), alpha (patch), and beta (habitat turnover) diversity has been highlighted in recent conceptual papers (Ward 1998, Ward and Tockner 2001). However, there is a limited pool of empirical data supporting this, particularly in relation to the rarity and conservation status of different groups of organisms. In the United Kingdom, the conservation potential of ERS habitat for rare invertebrates has been systematically assessed and Hammond (1998) estimated that 3.5 percent of the total British beetle fauna are riparian specialists. Fowles (1989) illustrated the potential importance of 'shingle' (fine gravel) ERS in providing habitat for a wide range of rare and nationally scarce species of Coleoptera, and the UK Biodiversity Steering Group has created grouped-species action plans for six species of ERS Coleoptera, two species of Diptera, and one species of water beetle (Anon 1995) that are seen as ERS specialists. Recent work on ERS beetles on rivers in England and Wales recorded 81 rare beetles species, of which 42 (52 percent) are considered ERS specialists (Sadler et al. 2004). In a parallel survey of ERS habitats in Scotland and northern England, Eyre et al. (2001a,b) recorded 115 species with conservation protection status. Elsewhere in Europe, active gravel-bed rivers are characterized by the occurrence of rare invertebrates species, such as orthopterans (Reich 1991) and there are 21 species of ground beetles on the German Red Data Book lists (Manderbach and Reich 1995, Plachter and Reich 1998). However, little systematic research has been aimed at establishing the extent and nature of the conservation resource in Europe and much work remains to be done.

Ecological dynamics

Established theories of river-floodplain interaction, such as the river continuum (Vanotte et al. 1980) and the flood pulse concept (Junk et al. 1989, Tockner et al. 2000), emphasize the movement of energy and nutrients from the floodplain to the river. Only the riverine productivity model (RPM) hypothesizes that much of the primary production is autochthonous, wrapped up in grazer-scraper and detritivore pathways (Thorp and Delong 1994). However, within these large-scale systems there are a number of smaller scale 'food web subsidies' that move across terrestrial and aquatic boundaries (Polis et al. 1997). There are growing numbers of studies that highlight the significance of terrestrial arthropods as essential food resources in aquatic stream ecosystems (Nakano et al. 1999, Kawaguchi and Nakano 2001), that is, a land to river transfer.

ERS systems, however, do not fit these general patterns, as they are characterized by a deficit of organic materials and are supported by a transfer of energy from the river (? or groundwater). As a result, the food webs appear beguilingly simple; they are predator-dominated with an abundance of ground beetles (Carabidae), rove beetles (Staphylinidae) (Eyre et al. 2001a, b; Sadler et al. 2004), ants (Formicidae), and spiders (Araneae), with very notable spatial aggregation along the shorelines on the edges of the ERS. Aquatic drift appears to be a major source of food for ground beetles (Carabidae) on braided systems in Alpine and upland rivers (Hering and Plachter 1997, Hering 1998) and spiders on gravel bars in New Zealand appear to gain more than 50 percent of their body carbon from aquatic sources (Collier et al. 2002). Indeed, spider biomass inhabiting stinging nettle stands along a river in Germany was found to be higher where aquatic insects (mainly chironomids) aggregated (Henschel et al. 2001). Similarly, Sabo and Power (2002) show how aquatic insects subsidize secondary consumers such as lizard populations and terrestrial invertebrates. At a basal trophic level, similar subsidies appear to operate linking stream algal production and the abundance and distribution of detritivores, such as grasshoppers, on braided sections of rivers (Bastow et al. 2002).

But species and population persistence are not merely a result of the availability of food resources. In highly dynamic environments, population persistence requires that individuals are adapted to disturbance events.

Species traits and functional ecology

For most natural floodplains, there is stochastic element in the seasonal flow pattern that is not predictable, and as a result floodplain species are characterized by having traits that are adapted to sporadic, unpredictable, and often high magnitude inundation events. Carabids and probably most other riparian arthropods are all capable swimmers (Andersen 1968, 1985) and it seems likely that passive downstream transport is an important form of dispersal, although no data are available to substantiate this hypothesis. Moreover, laboratory experiments illustrate that individuals of species of Bembidinii (a tribe of ground beetles) can survive inundation for up to 48 hours before large-scale mortality (Andersen 1968, 1985b), which is sufficient time to find landfall and suitable habitats during even the largest of flood events.

Desender (1989) records high levels of macroptery in riparian beetle species, a useful trait in an environment where inundation is a sporadic and unpredictable event. Indeed, the proportion of macropterous beetles on unvegetated ERS varies in relation to ERS location in the floodplain. Close to the river edge, between 91 and 99 percent of the species are capable of flight and this proportion falls to about 76 percent on individual habitat patches that are rarely inundated (Plachter 1986). Moreover, marked carabid individuals from an ongoing population study on the River Severn (Bates et al., submitted) have been found up to 1.5 km away from their initial point of release several weeks later, and after a large inundation event. Bonn (2000), in a study of the River Elbe in northern Germany, demonstrated not only that carabids fly actively after the spring and autumn floods, normally toward the river, presumably in search of newly deposited food resources, but also that species appear to move away from the river immediately before flood peaks.

Many ERS species have life cycles that are in tune with the seasonal flood in temperate regions. For example, a large proportion of riparian specialist carabids over-winter as first year imagines in habitats that are less susceptible to winter floods, such as high in trees or in grass tussocks higher up on the banks (Lott 1996), although some will travel considerable distances away from the river (Zulka 1994). Similarly, other ERS species, such as the large Lycosid spider, *Arctosa cinerea*, which is widely distributed in Europe, hibernates higher up the banks in habitats less like to be flooded during winter (Framenau et al. 1996a,b). There are some notable variants to this general scheme, however. For example, a large carabid, *Nebria picicornis*, of Alpine rivers usually hibernates as a larva and the adults, which can more easily avoid inundation, emerge early in summer, prior to the period of peak snow-melt (Framenau et al. 1996a, Manderbach and Plachter 1997). However, some individuals of *N. picicornis* overwinter as adults providing an exceptional degree of phenological plasticity, which is thought to be a mechanism for dealing with unpredictable environments (Plachter and Reich 1998). Although, limited data exist to suggest a high level of adaptation in floodplain systems, there is a clear need for models that couple hydrological and ecological dynamics at the level of individual populations before one can consider the likely outcomes of changing hydrological regimes as a result of longer term climate changes or catchment modifications.

Population dynamics

The highly dynamic nature of ERS environments suggests a high turnover of habitat patches. Recent work on the Tagliamento River in Italy illustrates that under natural conditions floodplain habitats are extremely dynamic, even over short time scales (van der Nat et al. 2003), but the configuration of habitats remains relative stable (Arscott et al. 2002), providing a continuity of habitats that are available for colonization.

The high turnover of ERS habitats necessitates the transfer of some individuals between habitat patches, so it is unlikely that populations of ERS specialists exist as ‘separate’ populations. Population types range from those that function as ‘sources,’ ‘sinks,’ and ‘pseudo-sinks’ (Pulliam 1988, Pulliam and Danielson 1991, Watkinson and Sutherland 1995) and ‘mainlands’ and ‘islands’ (MacArthur and Wilson 1967), all of which may exist as ‘metapopulations’ or as single ‘patchy’ populations (Harrison 1991). Although it is difficult to map individual populations to any particular type, populations can be classified along two axes: a ‘mobility’ axis, which describes the level of dispersal between subpopulations, and a ‘compensation’ axis, which describes the degree to which a subpopulation exports or imports individuals (Thomas and Kunin 1999). In short, the structure of the population relates not only to the stochasticity of birth and deaths and immigration–extinction, but also to patch size and location. A few previous studies of ERS specialist invertebrates have demonstrated some degree of spatial structuring of the populations (Reich 1991, Manderbach and Reich 1995). Using a metapopulation model for the grasshopper *Bryoderma tuberculata*

Fabricius, Stelter et al. (1997) illustrated that large numbers of shingle bars are needed in a given catchment to support populations of this mobile invertebrate because females have limited dispersal abilities and older, higher, and more stable shingle bars, which were less prone to inundation, provide sources from which dispersal can take place after severe flood events. The population structure of this species is akin to a classic 'metapopulation,' situated low down on the mobility axis of Thomas and Kunin (1999). In contrast, research on the ground beetle *Nebria picicornis* (Fabricius) showed that most individuals were recaptured more than 400 m and some more than 800 m from their release point (Manderbach and Plachter 1997), suggesting the potential for a 'patchy' population structure.

As many ERS invertebrates appear to have very narrow habitat requirements, it is possible that ERS metapopulations will not only relate to landscape structure, but also be a function of habitat quality. Fleishman et al. (2002) suggest that for populations of the butterfly (*Speyeria nokomis apacheana)* measures of habitat quality (mainly derived from vegetation composition and quality) explained more variation in occupancy and turnover than patch geometry measures such as area and isolation. However, Moilanen and Hanski (1998) found that the use of additional environmental data on habitat quality failed to improve significantly the fit of a metapopulation model for the Glanville fritillary butterfly (*Melitaea cinxia*) beyond that which would be expected from one based on the effects of habitat area and isolation alone. The potential affect of habitat quality on the colonization and persistence ERS invertebrate populations has yet to be addressed.

Hydroecological dynamics

Figure 9 provides a schematic that identifies the key elements of an ERS environment. Box A identifies the important balance between the hydrological variables that cause habitat destruction and creation and those that lead to stabilization. Box B considers the key ecological elements in the system and how they interact with important physical habitat properties (Box C). For braided ERS systems, the intensity and frequency of disturbance events that reconfigure habitat patches is a central and important aspect sustaining floodplain biodiversity (Ward 1998, Ward and Tockner 2001). The temporal-spatial variability leads to longitudinal (Framenau et al. 1996a), lateral (Bell et al. 1999, Bonn et al. 2002), vertical, and temporal patterns in the biota (Ward 1989, 1998; Ward et al. 2001, 2002). At a basic level the system is best viewed as a process of patch creation and destruction resulting from the balance between the rejuvenating force of flooding events and ERS stabilization due to vegetation succession.

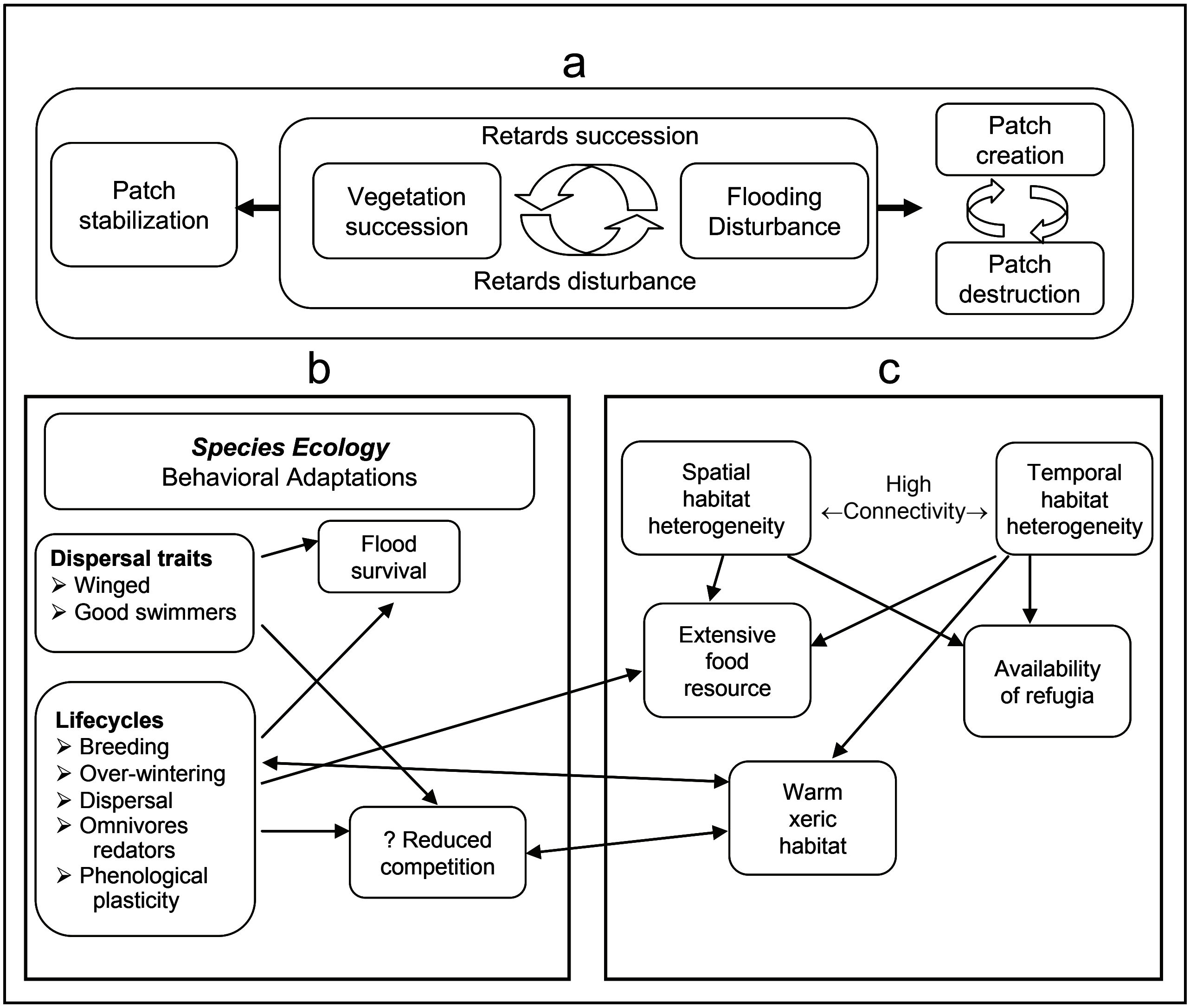

Figure 9. The main elements of a hydroecological model of ERS habitats

River regulation can enhance vegetation succession and reduce habitat availability. Thus, on the River Isar in Germany, downstream of the Sylvenstein dam, the amount of bare and pioneer-vegetated ERS habitats decreased by around 80 percent between 1925 and 1985 (Plachter and Reich 1998) and Von Manderbach and Reich (1995) found that the number of endangered carabid species inhabiting the downstream reach was only 2, while 15 were found in reaches above the dam.

Modeling Change

ERS specialist species should increase in diversity. Many have traits and preadaptations for highly disturbed environments, so that as disturbance increases then these species should increase in diversity. Most ERS species should be able dispersers, but data concerning population structure and dispersal potential are very limited. However, a large number of ERS specialists have low abundance and limited distributions in Europe. The work carried out so far in the United Kingdom suggests that species that are ERS specialists and rare ERS species are most abundant at intermediate levels of disturbance, where habitat diversity is maximized (Figure 10). However, no studies have attempted to relate or model these elements of species diversity directly to river flows.

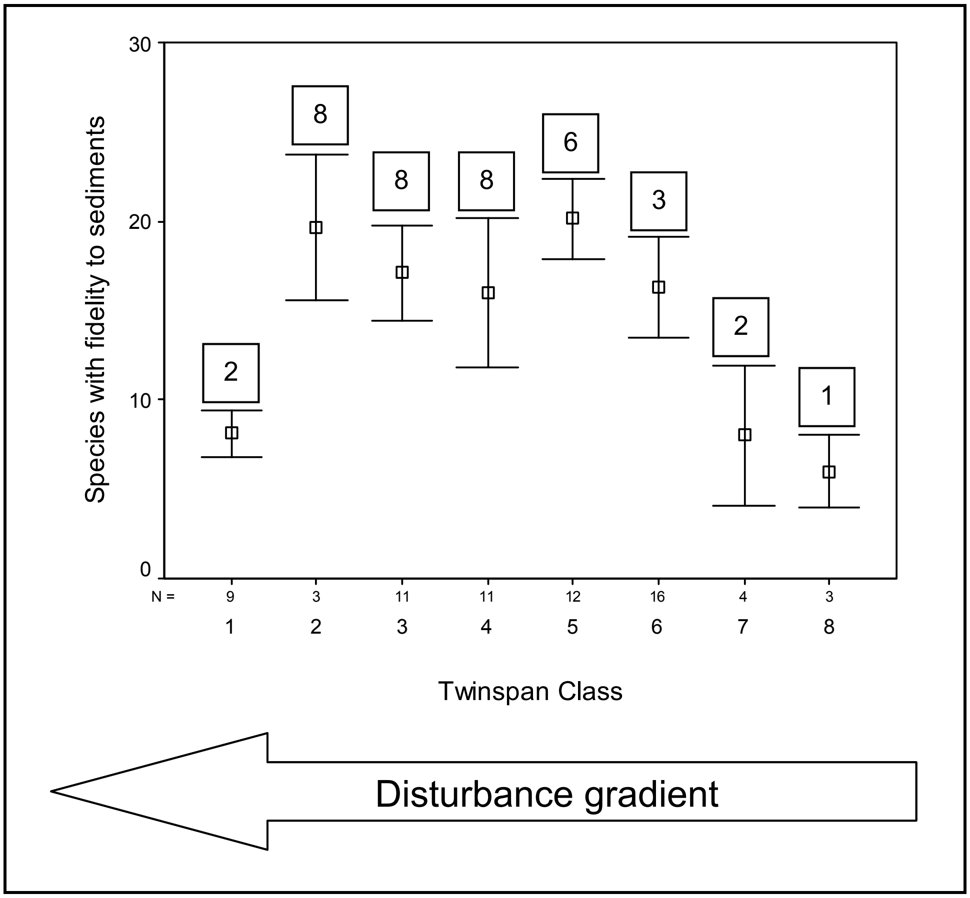

Figure 10. ERS species diversity, indicating the number of rarities, across a disturbance gradient that encompasses 82 sites in England and Wales. The axis shows groupings of sites using TWINSPAN. The numerals relate to the number of species found in each class with UK conservation status. Data from Sadler et al. (2004)

Flow variability

The basic guiding principles concerning the influence of changing natural flow regimes on aquatic biodiversity (see Chapters 1 and 6) were designed with aquatic species in mind, but are relevant to ERS species (Table 4). There are a number of empirical studies that confirm the importance of patch variability and habitat complexity and population dynamics in floodplain environments and a larger body of research examining aspects of ERS species lifecycles and seasonal patterns (Principles 1 and 2; Table 4). However, work concerning lateral and longitudinal variation in riparian invertebrate assemblages (Principle 3; Table 4), emphasizes the importance of lateral and longitudinal habitat changes in relation to inundation,. Principle 4 remains untested.

Table 4
ERS Invertebrate Communities and Bunn and Arthngton's (2002) Guiding Principles

Principle	Description	Biological response	Sources
1	Channel form Habitat complexity Patch creation and disturbance	Variations in species diversity and rarity Population structure	Manderbach and Reich 1995, Sadler et al. 2004, Stelter et al. 1997, Reich 1991, Bates et al. submitted
2	Seasonal predictability and life cycle synchroneity	Phenological plasticity Space partitioning Temporal partitioning Dispersal – food Dispersal – flood	Manderbach and Plachter 1997, Framenau et al. 1996a,b, Bonn 2000, Andersen 1968, 1969, 1978, 1979, 1983a,b, 1985a, b, 1988
3	Lateral and longitudinal connectivity essential to viability of populations	Longitudinal and lateral variation only	Framenau et al. 2002, Sadler et al. 2004, Bell et al. 1999, Bonn and Kleinwächter 1999, Bonn et al. 2002,(Manderbach and Reich 1995
4	Natural flow regime discourages invasions	No data on invertebrates	Untested

It seems possible to relate the magnitude of events with thresholds in the ecological system. It is clear the large-scale events may provide a *resetting* or *rejuvenating pulse* that causes whole scale reconfigurations of channel morphology. The impact of such an event in the short term may be increased mortality for many organisms, although in the longer term increased habitat availability should lead to enhanced levels of biodiversity as species of plants and animals recolonize the 'new' habitat. One might also identify a lower threshold event, or *cleansing pulse*, which corresponds to the 'flow pulse' identified by Tockner et al. (2000). These lower magnitude events may prove to be essential for ecosystem maintenance insofar as they inhibit vegetation succession and lead to spatial and temporal shifts in food resources. As long as the flows are sustained between these two thresholds, one might predict that the system would be resilient to change. It seems likely that variations in the frequency of flow events could lead to similar outcomes for ERS specialists, depending on their timing and seasonality.

Flow duration

The importance of flow duration for ERS invertebrates is not well documented. Some species have behavioral traits that permit them to 'ride out the storm' either in situ or ex situ (Andersen 1968, 1969, 1983a, b, 1985a). However, the timing of high flows may be of particular significance. Particular species have life stages that are less able to cope with inundation in mid- to late summer when the larvae are active on ERS. Large events at this time could cause extensive mortality and have some implications for recruitment in the next year.

Conclusion

ERS habitats are unique environments globally. Not only are the active gravel-bed rivers that create them under continued threat from regulation, catchment modifications, land use changes and so on, but they exhibit high productivity and are home to a well-adapted and often rare group of organisms. The flood is the engine that drives ecological diversity, by cycling nutrients, providing food, and constantly regenerating habitats. Population dynamics of the few invertebrates that have been studied intensively suggests patchy populations with some spatial structuring related to habitat size and inundation frequency. It seems likely that the latter is the key structuring variable that creates the appropriate habitat quality required by ERS species.

There are very few autecological studies on ERS species and even fewer population studies. Perhaps more significantly, we currently have a poor understanding of key ecological processes such as competition, food web dynamics, energetics, and the relationship between productivity and species diversity (Loreau et al. 2001). Thus, a number of key questions emerge: How threatened are these systems? Can we define more ecological meaningful hydrological variables? Are there any thresholds in the hydroecological system? Do certain habitats (e.g. large, stable bars) within the mosaic provide nodal points that are important source habitats? How are these related to biotic changes (e.g., mortality, patch extinction, dispersal)? How important are biotic interactions (e.g., food web, competition, energy flows, and trophic relations)?

12 Vegetation Patterns and Ecological Dynamics Along Narrow Riparian Zones

Introduction

As ecotones between aquatic and terrestrial systems, riparian zones demonstrate several specific ecological properties (Sedell and Froggat 1984, Salo et al. 1986, Naiman et al. 1988, Holland et al. 1990, Gregory et al. 1991, Malanson 1993, Naiman and Decamps 1997, Tabacchi et al. 1998, Tockner et al. 2002). In such transitional systems, vegetation plays a core role as both ecological structure and processor. The view of riparian areas as “green strips” along rivers has become popular but these riparian areas are usually considered to offer poor recreational value, especially in Europe. However, demonstration of their ecological functions has increased their value at both political and popular levels. In particular, the high potential of riparian buffers to reduce nonpoint source pollution has stimulated applied ecological research (Haycock et al. 1997).

A common assumption is that riparian width is the most important criterion for estimating ecosystem performance or health. It is generally assumed that the relative importance of a riparian zone with respect to channel size (width) reflects its ecological potential, whereas some studies emphasize the importance of boundary development as a key-indicator for many processes. Especially in the temperate zone, river systems have become more and more fragmented, narrowed and straightened following management and land-use changes. A growing body of evidence indicates that narrow vegetated riparian strips do not promote a coherent ecological functioning. The question of how wide a riparian buffer should be is still in people’s minds since efforts to restore self-sustaining (i.e., wide) riparian zones are costly and time consuming. This contribution aims to identify the implications of riparian structure for the role of vegetation in ecological dynamics and to provide a management perspective.

Delineating Riparian Zones

The river continuum (Vannote et al. 1980), flood pulse (Junk et al. 1989), serial river discontinuity (Ward and Stanford 1995), and nutrient spiralling

(Newbold et al. 1982) concepts have clearly highlighted the prominent role of riparian zones in the functioning of river systems. The definition of riparian zones as distinct ecosystems has induced several debates during these last decades (Johnston and Naiman 1987, Holland 1988, Welsch 1991, Gregory et al. 1991, Naiman and Decamps 1997, Malanson 1993). In natural conditions, the biophysical characteristics of riparian vegetation represent efficient indicators of the spatial extent of riparian systems, as it integrates most riparian ecological functions. Riparian vegetation is most often distinguishable from the surrounding terrestrial matrix as "green strips" containing mainly species adapted to high moisture content and to hydrological and physical disturbance. In some cases, however, the structure or the nature of riparian vegetation can hardly be separated from the adjacent terrestrial patches. This is the case, for example, when swamps or groundwater-fed wetlands board the river (Stanford and Ward 1993), when steep slopes occur along constrained channels, where xerophilic vegetation grows on alluvial levees, and where the natural vegetation is fragmented by human pressure (agriculture) or enlarged by plantations.

The gradient of inundation may be the most objective and strongest integrator of riparian influence. The gradient of inundation by surface waters is an obvious parameter of influence. In their attempt to model internal processes in riparian zones, Gold and Kellogg (1997) pointed out the need to recognize watertable dynamics as a full component of the riparian model. Thus, considering that groundwater and surface water dynamics are the main controls of the riparian ecosystem, we propose to delineate an indicator variable from hydrological data series (Figure 11). This very simple model defines the space of interaction between nonatmospheric water and substrate as a gradient of probability of inundation of both superficial area (F.Z.) and unsaturated groundwater zone (U.Z.). Swamp zones occur when the U.Z. overlaps the F.Z. TWD defines the coupling between surface and ground waters. The model can be coupled to an D.E.M. (Digital Elevation Model) to produce a map of the riparian zone.

When narrow/linear riparian zones result from human activities, two theoretical cases should be considered. The first assumes that human impacts affect hydrology. In this case, all the parameters in the model illustrated in Figure 11 are modified, leading to irreversible ecological patterns and dynamics. The second case assumes that hydrological properties are not modified (and hence, the delineation of the riparian zone), and then the expression of ecological patterns and processes are only restricted by resource limitation (e.g., forest fragmentation, nutrient input…). Short-term recovery would be expected with pressure release.

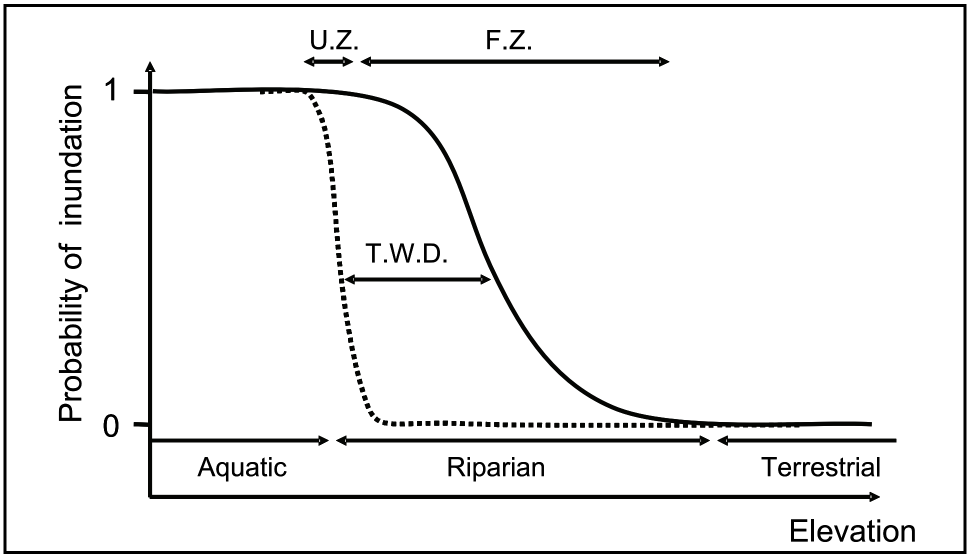

Figure 11. Hydrological representation of the riparian zone as a sum of transitional gradients. Dotted line: long-period probability of inundation by groundwater as a function of elevation. Solid line: same, for river water level. U.Z.: unsaturated zone of the water table; F.Z.: flooded zone domain; T.W.D.: transitional water table distance (physical difference in elevation between the inflexion points of the two curves). Riparian zone is defined as the common domain of the 95 percent confidence intervals for the two cumulative distribution functions. Note that transition curves can be asymmetric

Constrained Riparian Zones

Narrow or linear riparian strips can occur under natural, pristine, or subpristine conditions. This is particularly the case for headwater streams, along which geomorphologic (bedrock, coarse sediments, V-shaped valley, thin aquifer) and biogeographic conditions do not allow a substantial development of a riparian corridor. Owing to the linear development of low-order streams, this category might be significant at the basin scale when taking the cumulated area/boundary into account. Although less frequent, bedrock or naturally incised channels may occur in large lowland river systems, limiting the extent of the riparian areas. This is also the case along most semiarid or arid rivers where the riparian area is limited to a narrow strip. Despite their narrowness, naturally confined riparian zones usually exhibit high ecological potential. On one hand, headwater streams develop most of the aquatic-terrestrial interface at the watershed scale and are sensitive to upland runoff. On the other hand, the high contrast between lowland confined riparian zones and the surrounding matrix makes them intense ecological attractors for ecological communities and processes.

Consequences for Biodiversity Patterns and Processes

Depending on their origin (river regulation or land use change), narrow riparian strips may have different consequences on biodiversity-related processes. The corridor structure of natural riparian zones usually includes a complex mosaic of patchy habitats reflecting the interaction among substrate heterogeneity, water availability, and plant succession. Because of hydrological disturbance, the successional gradient developing from the waterline to the floodplain edge is not as regular as the one found along lakeshores. Riparian zones tend to exhibit continuous and homogeneous patterns along poorly disturbed watercourses, especially along canals and regulated streams. This pattern is also observed along headwater streams. However, in natural conditions, most habitats—especially earlier stages—occur as distinct patches. Natural habitat fragmentation tends to decrease with higher successional ages, although the apparent homogeneity of riparian forests hides a rather high level of topographic, substrate, moisture, and even floristic (understory) heterogeneity.

As the average size of the patches becomes higher, "interior" species may indicate within-patch stability in ecological conditions. These species are usually more sensitive to changes than "edge" species. In comparison to wide riparian zones, narrow riparian zones are expected to exhibit sharp, or incomplete, successional gradients. The patchiness and diversity of the habitat mosaic are lower. When narrow riparian vegetation strips are artificially maintained under hydrological disturbance, the riparian mosaic may represent a truncated successional gradient (early or old stages, depending on the exposure to disturbance).

Habitat diversity and turnover have direct consequences on biodiversity (Naiman et al. 1993, Planty-Tabacchi et al. 1996). In riparian zones, higher levels of biodiversity are expected in middle courses, where successional gradients develop at their maximum, and where habitat turnover is maintained at high levels as a consequence of intermediate disturbance regimes (Decamps 1996, Naiman and Decamps 1997, Pollock et al. 1998, Tabacchi et al. 1998, Middleton 1999). Habitat-rich mosaics provide not only suitable environments for distinct plant and animal communities, they also contribute to sustain high interhabitat dispersal rates and metapopulation processes by maintaining fragmented, but frequent, and poorly isolated patches of the same habitat. The internal resilience attributable to these properties is likely to be disrupted by longitudinal fragmentation of the mosaic. Physical fragmentation may be balanced by hydraulic connectivity for numerous organisms depending on dispersal by water (Andersson et al. 2000a). It may also initiate certain favorable conditions for invasions of exotic or native nonriparian species (Tabacchi and Planty-Tabacchi 2001, Planty-Tabacchi et al. 2001). We have some evidence that natural hydrological regimes allow both high invasion rates and high levels of biodiversity (Planty-Tabacchi et al. 1996), whereas river regulation (including narrowing and fragmentation of riparian zones) leads to pauci-specific invasions, with dramatic consequences on biodiversity (De Waal et al. 1994, Nilsson and Berggren 2000).

Narrow riparian zones increase the risk of fragmentation and of reduction in habitat diversity. However, during a study of the use of riparian strips by small mammals in boreal regions, Darveau et al. (2001) have shown that some species prefer larger strips whereas others prefer narrow strips. One conclusion of this study is a recommendation to promote longitudinal and transverse heterogeneity for biodiversity management. Studying the dispersal potential offered by riparian strips in Illinois, Burbrink et al. (1998) demonstrated that, surprisingly, wide areas of riparian habitat did not support greater numbers of species of reptiles and amphibians than narrow areas. The authors concluded that simply relying on easy-to-measure parameters, such as corridor width, is not enough to sustain riparian corridor effect on biodiversity, especially for species with low vagility. Kilgo et al. (1998) concluded that even narrow riparian zones can support an abundant and diverse avifauna, but that conservation of wide riparian zones is necessary to maintain the complete avian community characteristic of bottomland hardwood forests in South Carolina. As for terrestrial fauna, riparian zone structure may affect the diversity of in-stream animals, by providing specific food and temporary refuge, or by modifying the habitat conditions within the channel (shading, water temperature, etc.). Morris and Corkum (1996) illustrated the effect of riparian forest width on freshwater mussels community structure, but, it was not clear whether community structure was directly related to width, or to associated vegetation types.

Management Perspectives

Consideration of riparian zone width emphasizes the need to link physical processes (mainly, hydrological and geomorphological) and biological processes (physiological activity, species succession, and dispersal) in models with a landscape perspective. Depending on the surrounding context (land use, regional species pool, stream dynamics, and water regime) and depending on internal structure (spatial continuity and heterogeneity, hydrological permeability, vegetation dynamics), narrow riparian zones may have singular ecological processes or may show high ecological effectiveness in buffering and dispersal functions. The high ecological potential of naturally narrow riparian zones of headwater streams is now widely recognized, although many processes remain poorly understood. The conservation of riparian zones along low-order streams represents a major issue for effective management at the basin scale. This perspective is different for large lowland streams, usually associated with large floodplains and wide riparian areas.

The study of riparian processes involves the integration of both lateral and longitudinal hydraulic connectivity and includes the groundwater compartment. For many biogeochemical processes, however, the role of the unsaturated zone remains poorly understood. Whereas seasonality is taken into account in current research, local events (rainfall, erosion, etc.) are suspected to play major roles in initiating biogeochemical processes or in inducing various pathways. Water residence time in the different compartments (surface, soil, watertable unsaturated zone, groundwater, etc.) is also a key factor for many physical, biological, and chemical processes (McGlynn et al. 2003). This factor directly depends on substrate heterogeneity and on above- and below-ground vegetation structures, and in some ways may explain the significance of riparian zone width for geomorphological and biogeochemical processes. The role of leaf litter and other plant debris inputs as a favorable factor for plant and animal recruitment has been pointed out by several authors (Nilsson et al. 1999, Goodson et al. 2001). At the moment, except for some obvious patterns (e.g., leaf fall), seasonal and mid-term vegetation dynamics are still infrequently linked to such processes in ecological models.

There are a number of key areas to be advanced before river landscape management can be realized.

a. The role of vegetation as a source of DOC for microbial activity is now obvious, but we do not know the relative importance of locally borne carbon versus that exported through surface and ground waters. Whereas leaf litter input in small headwater streams is recognized as a key factor for CPOM and DOC supplies in the aquatic system, little is still known about the role of large woody debris as a slow-decomposing carbon supply. Also, almost nothing is known on the importance of the litter of herbaceous species drifted along large rivers during floods.

b. Little is still known about the role of spatial and taxonomic plant diversities on ecological processes. In particular, we do not know if critical thresholds exist for functional resilience and redundancy as a function of biological (plant) diversity. In other words, is biological complexity able to compensate for losses in riparian zone width in terms of ecological effectiveness? In turn, is artificial physical complexity (i.e., substrate or topography reconstruction) able to compensate for losses in biodiversity or in sinuosity?

c. Very little is still known about the critical thresholds in width for inducing changes in community structure and dynamics and, in turn, in the response of biogeochemical processes to these biological changes. In particular, the effects of timing of disturbance events throughout a wide range of biogeographic conditions have been poorly studied (Welsch et al. 2000, Smits et al. 2000, Hughes et al. 2001).

Current research on the role of propagule banks as a resilience factor for plant succession and as a natural tool for restoring riparian plant biodiversity from hotspots may provide an optimistic perspective for the management of regulated systems (Malanson 1993, Abernithy and Wilby 1999, Middleton 1999, Goodson et al. 2002). This could be a crucial point when considering metapopulation functioning, plant invasions, and community genetics with respect to spatial connectivity.

As ecotones, riparian zones (and especially the narrow ones) are expected to be extremely sensitive to direct (climate) or indirect (land use, water use) effects of long-term changes. Most riparian functions are expected to be altered while community structures have already changed (Mooney and Hobbs 2000, Nilsson and Svedmark 2002, Pinay et al. 2002, Tabacchi and Planty-Tabacchi 2003) as a result of both reciprocal interaction and direct human influences. Thus, global change remains also a key-point in the riparian perspective.

13 A Commentary on Advances in Modeling the Effects of Vegetation on Flow, Sediment Transport, and Morphology

Introduction

In the planning and design phase of river rehabilitation projects, the use of predictive models can aid in assessing and quantifying the expected outcome. Models that can be applied in this phase are numerical models for hydro- and morpho-dynamics to assess the future abiotic conditions. One-dimensional numerical models have been applied mainly for this purpose. At present, two- and three-dimensional models are becoming more common in river applications. Two-dimensional models better represent the river topography, the flow fields, and the distribution of erosion and sedimentation. Three-dimensional models more accurately model typical hydrodynamic phenomena, for example those that are ascribable to vertical density gradients. A major challenge is to model the interaction of vegetation with flow and morphodynamics.

This paper summarizes three case studies of the application of one-, two- and three-dimensional numerical models to simulate the interaction of flow, vegetation, and morphodynamics. In principle, this interaction is two-fold. First, river morphology determines factors that control the suitability of locations for specific vegetation to grow and, secondly, the vegetation itself affects the flow and transport of sediment, thus changing its environment. One of the first to extensively describe the floodplain vegetation dependence on abiotic factors was Dister (1980). Some more recent studies include Edwards et al. (1999), Bendix and Hupp (2000), and Baumgärter and Grüneklee (2002). The effects of vegetation on flow, sediment transport, and geomorphology are of interest to modeling abiotic aspects. Important papers include López and García (1998), Gran and Paola (2001), and Murray and Paola (2003). The interaction between both vegetation (and other organisms) and geomorphology is termed biogeomorphology. Recent papers include Brown (1997), Hughes (1997), Wolfert (2001), Brooks and Brierley (2002), and Gurnell and Petts (2002). This note focuses on modeling the effects of vegetation on their physical environment, i.e., on flow, sediment transport, and morphology.

One-Dimensional Modeling of the Rhine River

This first case study deals with the modeling of flood levels affected by vegetation development and floodplain sedimentation within a 50-km section of the Rhine River in the Netherlands. The objective of this case study is to assess the effects of cyclic floodplain rejuvenation measures on flood levels and biodiversity (Baptist et al. 2004). Cyclic floodplain rejuvenation aims to increase the flood conveyance capacity, and the biodiversity of floodplains, by mimicking natural rejuvenation processes.

The study used the SOBEK modeling suite for one-dimensional analyses. Floodplain sedimentation and the development and succession of floodplain vegetation were modeled with two separate rule-based models applied in a two-dimensional GIS delivering input to SOBEK. The sedimentation model described the sedimentation rate per inundation day, for various morphological floodplain units. The vegetation model described the succession of vegetation dependent on the local conditions. The model simulation described the effect on the flood levels in the Rhine River of the combination of increasing hydraulic roughness and sedimentation of the floodplain over a period of 50 years. In case the computed water level exceeded the design level for safety in a floodplain section, cyclic floodplain rejuvenation measures were implemented during the simulation. These measures consisted of (combinations of) the removal of floodplain forest and the removal of sediment.

The study showed that the strategy of cyclic floodplain rejuvenation was able to sustain safe flood levels when about 15 percent of the total floodplain area was rejuvenated with a return period of 25 to 35 years. It also showed that these measures increased the diversity of floodplain habitats. One-dimensional analysis clearly demonstrated that the natural succession (landform and vegetation) in floodplains can diminish flood conveyance through a reach and that, without frequent resetting mechanisms, the flood safety cannot be guaranteed. However, this study also stressed the need for better predictive modeling of the interactions among vegetation, flow, and morphology, preferably in two- or three-dimensional models.

Two-Dimensional Modeling of Secondary Channels in the Waal River

The second case study deals with a complex of three man-made secondary channels that were created as part of a river rehabilitation project of the Rhine floodplains. The objective of this case study is to assess the morphodynamic behavior of these secondary channels under different conditions of vegetation development on the channel banks (Baptist and Mosselman 2002).

This study used the Delft-3D modeling suite applied in a two-dimensional morphodynamic mode, together with a rule-based vegetation model that describes the development of floodplain vegetation. Figure 12 presents a schematic diagram of the model relationships applied in this case study. The effect of the vegetation on the transport of water and sand was modeled in the common way, as an increased bed roughness. The sediment transport formula applied assumed uniform sediment with a median grain size of 300 µm. The model described the morphological developments over a simulation period of 30 years. In one scenario the hydraulic roughness of the floodplain remained constant, whereas in the other scenario the hydraulic roughness changed because of vegetation growth and succession.

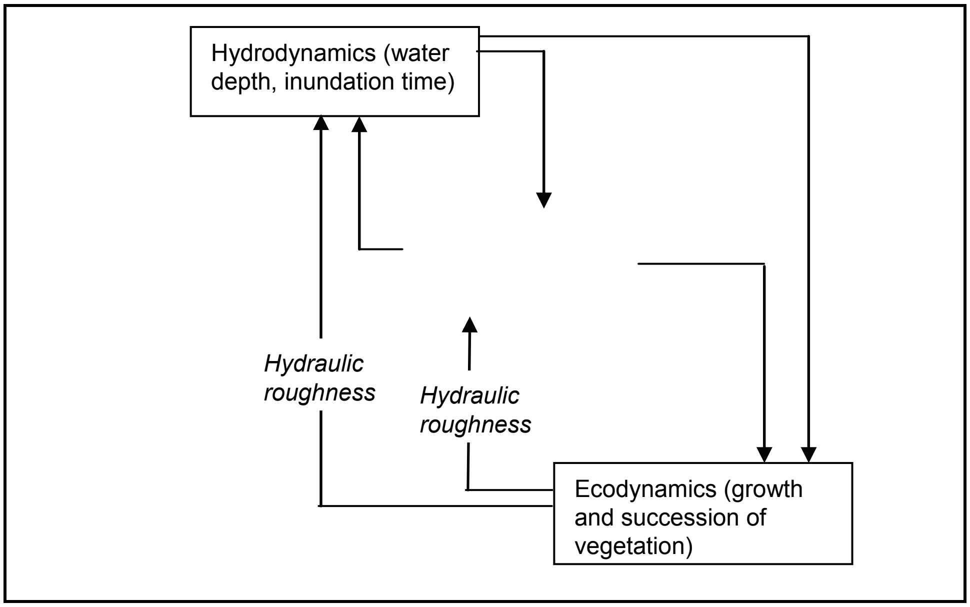

Figure 12. Relationships between the model components used in the Waal River Study

This case study showed that the medium-term morphological development of the secondary channels was affected by vegetation development. Generally speaking, there are remote and local effects of vegetation on morphology. The remote effects on the channel bed of the secondary channels were: (a) near the entrance of the largest channel, the bed eroded, as observed in reality, and (b) erosion was more pronounced in the runs with vegetation development, because then the increased roughness of the banks pushed the flow toward the channel axis. The mean erosion rate at the entrance was increased by 40 percent owing to the development of vegetation. The remote effects of vegetation on the morphodynamics of the smaller channels showed that sedimentation increased by 25 to 70 percent owing to the development of riparian vegetation. The magnitude of sedimentation agreed with final measurements. The local effects on floodplain levels, however, were erroneous. A couple of small islands that were present in the initial bed topography were washed away in the morphodynamic simulations. This local effect is a model flaw. In reality the vegetation stabilised the islands, but in the model the vegetation enhanced the transport of sediment. Furthermore, the simulations show a large eroding area downstream of a sand quarry. This resulted from the trapping of sediment in the deep pit. Moreover, the presence of vegetation enhanced the local transport capacity in the simulation, leading to an exaggerated erosion.

Three-Dimensional Modeling of the Allier River, France

The third case study deals with a natural section in the River Allier. The objective of this study was to model the influence of vegetation on the flow velocities and morphology in this area. The Allier River is chosen as a reference river for the future Border Meuse in the Netherlands, after its rehabilitation.

A special version of Delft-3D, in a 3-D hydrodynamic mode, was applied, with the vegetation modeled as rigid cylinders. A digital elevation model of the study area was acquired with the aid of RTK-dGPS and leveling. Aerial photographs of vegetation were analyzed and vegetation types were mapped and subdivided in units that were characterized by cylinders with a certain diameter, height, density, and drag coefficient. In this case study, results for flow velocity and bed shear stress were compared for the 3-D model and the 2-D model with enhanced bed roughness. In the latter case the vegetation units were also expressed in terms of a roughness height.

Results of this case study showed significant differences in bed shear stress patterns between the approach with vegetation cylinders and the approach with enhanced bed roughness (Figure 13). In the approach with enhanced bed roughness, the bed shear stress increased along with the bed roughness and showed very high values in places with dense vegetation. In the three-dimensional approach with vegetation cylinders, the bed shear stress was considerably reduced in places with dense vegetation. The distribution of bed shear stress was modeled more reliably, which is an important prerequisite for modeling the effects on morphodynamics.

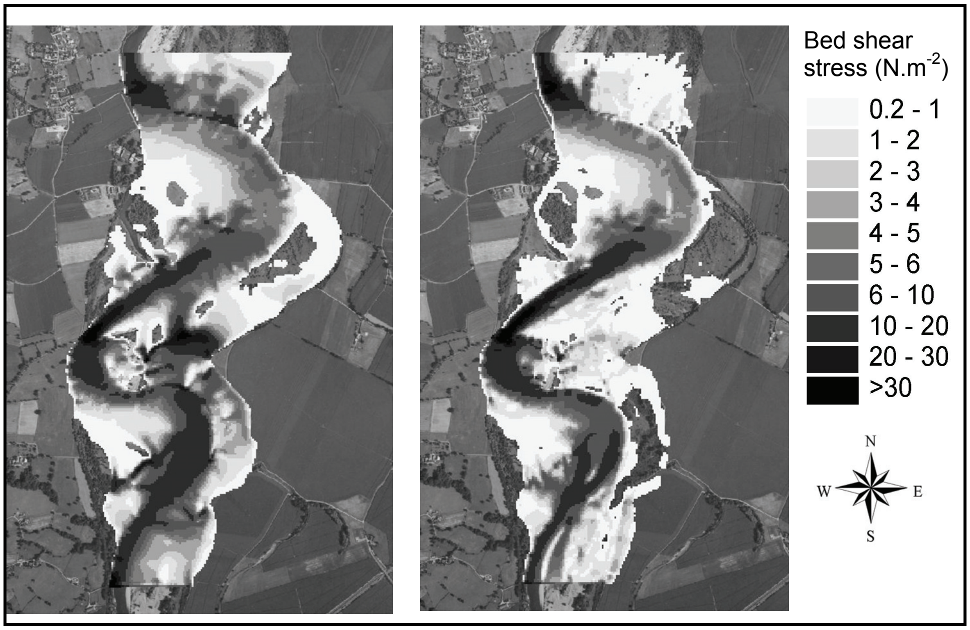

Figure 13. Computational results for the bed shear stress (left panel: vegetation modeled as bed roughness, right panel: vegetation modeled as rigid cylinders)

Discussion

The common approach to modeling vegetation roughness as enhanced bed roughness yields erroneous results in combination with common sediment transport equations, as was shown in the second case study. A modeling approach has been developed in which vegetation is schematized as thin, vertical cylinders that slow down the flow because of the drag force on these cylinders. Furthermore, the presence of these vegetation cylinders affects the turbulent energy and dissipation of turbulence. The processes that are modeled in this way are more physically based than the approach using increased bed roughness. As these processes are incorporated in a three-dimensional model, the flow and turbulence properties can be modeled over the vertical. The newly developed 3-D model is capable of describing more accurately the effects of vegetation on the bed shear stress, as was shown in the third case study. This forms a basic ingredient for modeling sediment transport.

Modeling vegetation as rigid cylinders has been tested against data on flow and turbulence properties obtained in flume experiments. These results show that, even for flexible vegetation, reliable results are obtained. Results of a flume experiment on sediment transport with flexible, submerged vegetation show that indeed the sediment transport is reduced by the reduced bed shear stress. On the other hand, the transport capacity for conditions with a relatively low bed shear stress is enhanced by an increase in turbulence between the vegetation stems. It is recommended that continuous effort be put into experimental, field, and modeling studies to understand the interactions of vegetation, flow, and sediment transport at the level of principal processes.

14 Integrating Human and Environmental Water Needs in River Management: A Summary of Workshop Discussions

Perspective

This Workshop brought together a small group of experts: physical and biological scientists, engineers, planners, and policy makers. Most offered a western European perspective, with a range of traditions and experiences but all founded in the long history of basin development and river regulation, in nearly four decades of practical experience in river rehabilitation, and in an evolving legislative framework that will lead to the implementation of the European (EU) Water Framework Directive that came into force in December 2000. Within the Directive, the fundamental tool for managing water resources to achieve 'good ecological status' is River Basin Management Planning. Plans must be adopted for all European waters by 2015.

The context for discussions was the belief that, despite having experienced a paradigm shift from 'control by construction' to 'management,' river managers still lack appropriate science to realize the integration of human and environmental water needs. Those managers concerned with water resources planning have a tradition based in human needs driven by the economic risks of drought, and founded upon flow duration statistics. Engineers struggle to design deterministic solutions for flood control and navigation improvement that have sustainable environmental benefits. Scientists strive to develop an understanding of ecosystem response to flow changes that is necessary for advancing tools to analyze the sustainability of alternative water resource management plans.

Accepting the controversial perspective that riverine ecosystems have legitimate water needs is a challenge for everyone. In the United Kingdom, one of the main goals of the Environment Agency's Vision (Environment Agency 2001) is to achieve the integrated and sustainable management of inland waters and the protection and enhancement of wildlife. Its role is to protect the long-term future of the water environment while encouraging sustainable development of water resources. In achieving this goal, the Agency must consider all users, the environment, and the views of nature conservation agencies and organizations. In the future, the requirements of the EU Water Framework Directive are likely to further expose our lack of applicable models and the inadequacies of our scientific understanding to develop such models.

The Workshop discussions focussed on the issues that need to be addressed to establish a unifying framework for advancing the scientific knowledge necessary to develop appropriate tools for water resources management. Two key themes evolved: (a) the need for scientists to focus on variability and (b) the need for a scientific framework that is shared by both scientists and managers.

To facilitate discussion on how to promote integration of human and environmental water needs, the group agreed on the following working definitions of a few key terms:

a. *Water resources.* Encompasses both the quality and the quantity of water in a system and includes the concept of ecosystem integrity.

b. *Sustainability.* The attribute allowing an ecosystem to provide services for the current generation without diminishing the capability of that system to provide services for future generations.

c. *Disturbance.* An ecological impact beyond ‘normal’ natural variability, related to the resilience of a system.

d. *Variability.* The natural variation in physico-chemical variables, notably flow, which is an essential driving force for creating temporal and spatial habitat heterogeneity and thus a prerequisite for natural biodiversity.

A Direction for Science

The scale issue

Issues that deal with scales, of space and time, and hierarchy were an overarching concern. River basin management must deal with a multitude of scales that range from patches and habitats for individual species to the landscape. Questions arise such as, “Is ecological integrity scale-dependent?” “If so, should we measure it?” “How do we measure it?” Perhaps “catchment assessment” should occur at multiple scales simultaneously? To develop an integrated measure of the quality or status of a river basin, methods for assessing characteristics of a stream reach, the local area draining into the reach, and the drainage area upstream from the reach are needed. For example, measures such as riparian corridor continuity are considered important at both the reach scale and the drainage basin scale, and may result in very different values at the different scales. Also, floodplain-channel interactions are widely accepted as critical on a reach scale, but are rarely evaluated by managers as important at the drainage basin scale. Fundamentally, we lack a detailed understanding of the importance of river corridor architecture across the range of spatial scales. This Workshop highlighted a number of research gaps or areas of weakness that focus on links between structure and function across space and time scales, and these are examined below.

Structure versus function

In any analysis of river systems, both functional and structural components of the systems should be considered. However, because of the lack of appropriate process-based models, managers have often depended upon structural indicators that provide surrogates for process. This is quite difficult to do in practice because the relationship between function and structure is often neither clear nor appreciated by managers. For example, critical biogeochemical processes may be "invisible" at the scales used to describe and manage river morphology. To allow managers to incorporate process in their evaluations, readily apparent *indicators* of processes and functions will need to be discovered and developed, but these should evolve from improved basic scientific understanding. This Workshop focused on a number of key areas for further research within both reaches and sectors.

At the reach scale, information is needed on (a) the influence of the number, size, and spacing of specific patch types (participants emphasized, for example, the importance of springs, ponds, and exposed sediment patches); (b) the roles of juxtaposed or coupled patch types (such as islands and ponds) and the processes responsible for these roles; (c) the importance of processes that reset succession and drive habitat turnover at rates typical of particular settings; and (d) the feedback effects of vegetation succession and floodplain patchiness on the hydraulics of overbank flows.

At the sector scale, information is needed on the feedback effects between reaches, on the importance of continuity versus fragmentation (including "beads-on-a-string" models), and structural diversity. This will require new quantitative knowledge of the dynamics of key processes such as surface water and groundwater interactions; sediment routing (input, transport, dispersion, deposition, erosion); organic matter production (and changing composition related to the importance of the different autigenic, allogenic, and anthropogenic sources), and utilization through food chains; and seed and propagule dispersal and vegetation succession. At this scale, it will also be important to be able to incorporate knowledge developed at the smaller, reach scale.

Reference rivers and assessment of integrity

Given the long history of river regulation, the advancement of research to address the above knowledge gaps requires consideration of the need for reference rivers against which human impacts might be benchmarked. Reference rivers can be used to mimic, compare, or predict, but there are numerous issues that need to be addressed. Theoretically, reference conditions can be defined as hydrological areas (catchments), linear features (river corridors), and structural areas (sites), and these may be defined at all scales. But is the establishment of reference sites feasible and is the concept truly viable within the constructs of river basin management?

The functions that parts of a river basin can be expected to perform are to some extent dependent upon, and certainly reflected by, typology. Consequently, and especially given the lack of deterministic, process-based models, the classification of river basins and of the territory within the basin has been seen as critical for the assessment of basin integrity. Traditionally, classification has been performed by several means, using catchment-scale measures such as climate, geology, altitude, and landuse; measures of scale and location within the drainage basin or network, and along the river continuum; and sector or reach-scale characteristics, including flow regime and channel pattern. Discussion of advances in river ecology led the Group to propose new indicators of structural integrity, especially applicable to sector- and reach-scale classifications, that appear to have increasingly clear functional qualities (Table 5). Supporting evidence for this selection is presented in the previous papers.

Table 5
Proposed Indicators of (Structural) Integrity Within Fluvial Hydrosystems

Structural measures:	Measures of variability with discharge:
Riparian zone width	Shoreline length
Width of active corridor	Connectivity between habitats
Number of islands	Area or number of channels
Refugia availability	Components of detrital organic matter budget
Number of ponds	
Relative number of species: colonizing opportunists competing species ubiquitous species	
Permeability of channel bed sediments	
Degree of surface – ground water exchange	
Floating organic matter composition	

Within Europe, the issue of ‘reference sites’ and river basin ‘integrity’ has become integrated with discussions about the need for the standardization of surveying and monitoring of river basins with the view to establish European-wide ‘benchmarks’ against which human impacts and progress toward restoration may be assessed. The EU Water Framework Directive will require catchment-level assessment, placing emphasis on ecological status and integrity.

Focus on variability

The introduction of structural measures that are dynamic, varying with discharge, such as proposed in Table 5, would represent a major step forward in assessing the integrity of riverine ecosystems. However, this in no way masks the need for more fundamental research on process variability. The pattern and variability of the three key abiotic factors—flow, temperature, and substratum stability—have been advanced as the primary drivers of lotic and floodplain ecosystems over a range of space and time scales (e.g., Petts 2000, Tockner et al.

2000, Milner et al. 2001, Gurnell and Petts 2002, Richards et al. 2002, Church 2002). Furthermore, the degree of predictability of flow variability has been hypothesized as a major determinant of the importance of abiotic versus biotic factors in regulating population and community processes and patterns (see Poff and Ward 1989). An evolutionary perspective suggests that a predictable disturbance regime will offer more opportunity for a functional assemblage to develop than an unpredictable one. Ward et al. (2002) showed that it is the natural disturbance regimes that drive habitat turnover, the expansion and contraction of resource gradients, and the potential for competitive exclusion among species. Clearly, understanding the complex effects of abiotic variations upon habitats, and the responses of species and biotic communities, and their interactions, over ecosystem-relevant time scales, provides a major challenge for the future.

Despite considerable progress over the past two decades (see Special Issue of *River Research and Applications* 19, 5–6, 2003) the management of flows remains embedded in expert judgment, supported by tools such as the In-stream Flow Incremental Methodology (IFIM), implemented within river basins using local applications of "Hands-off flows"[1] applied to abstractions and constrained by the legacy of historic water allocations (e.g., Petts 1996, Petts et al. 1999). New science is required to understand the behavior of ecosystems, not individual abiotic or biotic components, but there is a lack of long-term data on most key components to understand system dynamics.

Hydrological variability must be related to habitat and species turnover. It is important to understand the effects of both disturbance events and subsequent recovery processes, especially with regard to the ways in which landforms and communities age and change in their sensitivity to disturbance. Major floods can reset entire systems and even cause the switch between rheophilic and limnophilic species; moderate floods disturb and rejuvenate systems, and low floods can enhance recovery and succession. However, within each river reach, the frequencies associated with these three types of event are likely to vary with (a) time since the last major flood and (b) the level of human impact on the river corridor (e.g., Gurnell and Petts 2002). The impact of extreme high- and low-flow events is also likely to relate to the timing of the event in relation to species traits and life cycles. In this context, changes in temperature become important in determining the timing of key biological 'windows.' The biota of river corridors have a wide range of generation times, from many generations per year to one generation in several years, and many are particularly vulnerable at key times of the year. For example, exposed-riverine-sediment specialists are most vulnerable in mid-late summer when larvae are active on the sediments. It is clear that better data are needed if scientists are to develop the knowledge and then the tools for achieving integrated management for human and environmental water needs.

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[1] A term used by water resource managers in the United Kingdom to define the flow (or river level) threshold at which abstractions are switched off or cut back; equivalent to 'flow reservations' in the United States.

Episodic change toward longer-term trends

In the past scientists and engineers have attempted to model rivers as steady-state systems. Although rates of colonization by plants and animals may be rapid, key water-quality and geomorphological changes can require decades and in some cases centuries. Polluted groundwaters can take hundreds of years to pass through intragranular (cf. fissured) aquifers and rivers may continue to receive contaminants from polluted groundwaters for many years after the pollution source has been controlled or eliminated. Channel form responds to the flow regime but in many cases (e.g., along regulated rivers below dams) channel change and the restructuring of channel–floodplain architecture can require more than a century to establish a new quasi-equilibrium condition. Petts (1987) suggested that research and management should focus on transient states that can be defined over management time scales of one or a few decades.

With reference to channel changes along regulated rivers, Petts and Gurnell (in prep.) suggested that, for many rivers, channel adjustments require such long periods of time that the modeling of channel and physical habitat changes and the resultant ecological changes should focus on the spatial scale of the reach over time periods of 10-30 years. That is, effort should be directed to modeling the sequences of transient states that evolve in response to the actual series of flows and sediment loads in each reach over management time scales, rather than focusing, as in the past, on predicting ‘regime’ changes. In every case, it is important to understand, if not isolate, natural variations over the range of time scales and any underlying trends in response to changes of the fundamental hydrological controls. Such changes in these controls could be caused by changes in climate, land use, or flow regulation.

Developing a Common Science Framework

The Workshop discussion focussed on three requirements for the advancement of a common science framework to integrate human and environmental water needs in river management: understanding data requirements, appreciating the different traditions of the expert groups working on this problem, and improving communication not only between expert groups but also with all stakeholders.

Data needs

As suggested above, a major issue is the plea by scientists for more data and the claim by managers that they have information overload. Often, there is a disconnection between the scale at which managers and scientists need to work and the scale of the information available. From a practical perspective, too many data may appear intimidating when what is required for management is consistent, simple, defensible tools and approaches. The dilemma is that scientists need detailed data across a range of temporal and spatial scales but managers only need the data once they have been transformed into useable management information.

The data mountain myth. Too often, at least in management fora, it is concluded that the large amounts of data that undoubtedly exist can be mined to solve a host of unanswered questions, the problem being simply to find and gain access to the data. Many rivers are characterized by datasets collected as part of long-term monitoring programs comprising regular or periodic surveys and environmental monitoring to meet regulatory requirements and focused on describing trends in time or space. In reality, there are a lot of long-term monitoring data, mostly for variables that are easy to measure. However, there are few synoptic data collected for advancing the science that can be used to infer component interactions, functions, and processes. All assessments of river system integrity must be considered in the context of normal, natural variability. There are limited data on the scale of natural variations of habitats, populations, or communities; there are few coupled data to advance improved models of abiotic-biotic interactions; and there are even fewer data to define threshold conditions for biota.

The collection of data over regular time and space intervals as part of river management frameworks, while valuable, is often of limited utility for describing environmental processes at the resolution required to develop forecasting tools and predictive models capable of testing management scenarios. Event-associated processes often elude the sampling designs of regular monitoring programs so that synoptic, flow-associated sampling necessary to describe river processes is generally not available, particularly for extreme high flows or droughts and especially for large rivers. Therefore, some findings about rivers are “black box” in the sense that ecological processes cannot be monitored in “real time” and must instead be inferred from “post-event” forensic evaluations of the river. These attributes make the response of rivers to alternative management actions, particular at flow extremes, difficult to forecast. The information needed to predict the results of a particular management scenario on the integrity of the basin often requires experimental ‘cause and effect’ sampling. Data monitoring is at the heart of the EU Water Framework Directive and there is an opportunity for advancing data monitoring designs that could interface with scientific research objectives as well as meeting management-information needs.

Problems of large rivers. Particular problems arise in advancing knowledge of large, natural rivers. Large rivers are complex systems that reflect many different hierarchically structured processes occurring over a diverse range of temporal and spatial scales. It is inherently difficult to accurately associate changes in the characteristics of a large river with the blend of variables responsible for a particular change. In addition, deployment of instruments and development of comprehensive monitoring programs for large rivers can be challenging because large spatial scales discourage high resolution sampling. Furthermore, particularly in natural rivers, (a) high energy environments and mobile channels defy many sampling technologies, (b) unstable flow and bed conditions make maintaining regular sampling stations difficult, and (c) along deep, slow-flowing lowland rivers, high turbidity limits direct observation and the application of optical techniques.

Issues in ‘modern’ landscapes. In ‘modern’ landscapes, the regulated nature of flows within ‘trained’ river channels may make monitoring more practicable. However, an additional constraint to the progress of knowledge is the artificial nature of catchments with complex and continually changing land uses, and the artificially influenced water quality that is typically more turbid and nutrient-rich than natural rivers. In consequence, much of our understanding of the dynamics of large rivers is qualitative, or at best, semiquantitative, and often based upon more-or-less regulated systems. This is why scientists argue for greater research effort on 'reference rivers, reaches and sites' to better understand the links between structure and function, and the nature of feedback processes that sustain the important goods and services of natural riverine ecosystems.

Data for all. A large river is hierarchically structured so that tools to forecast its response must integrate across the scales used by the different disciplines that build tools to manage rivers. If these tools can be successfully developed, then the disciplines can be broadly separated into three tiers: (a) basic scientists who work at the level of "first principles" within their discipline, (b) applied scientists who attempt to build management tools using good science, and (c) resource managers that attempt to use the tools to manage or restore large rivers. Hierchically structured monitoring programs could be designed to collect information for basic scientists to improve their understanding of large rivers, to allow applied scientists to build better management tools, and, in turn, allow resource managers to better manage large rivers. This feedback is particularly important between the applied scientists and the resource managers that use the tools.

Conflicting traditions

The varying traditions and conventions used by the different disciplines that study rivers, as suggested above, further exacerbate the difficulties of developing a common science framework that can be used to advance forecasts of the response of rivers to management actions. The contrast in approach is reflected by advances in aquatic ecology during the last three decades that can be broadly separated into two approaches—synthetic versus engineering—each with its own distinct origin and history of development.

The synthetic approach. In the synthetic approach, scientists have attempted to understand the ecology of rivers at a holistic level and to describe how important riverine processes vary over time and space. Noteworthy conceptual advances have been discussed in the preceding papers and these include the River Continuum Concept (Vannote et al. 1980), the Flood Pulse Concept (Junk et al. 1989), the Patch Dynamics Concept (Townsend 1989), the Natural Disturbance concept (Resh et al. 1988), and the Natural Flow Paradigm (Richter et al. 1996a).

The engineering approach. In the engineering approach, researchers have tried to develop suites of tools that could be used to predict river stage, velocity fields, and bedform as a function of discharge, and to predict velocity and shear stress at multiple points within the river channel. Using well-accepted governing equations, engineers were able to develop tools to simulate the bulk flow of water at a resolution sufficient to route stage and various conservative and nonconservative constituents. Further refinements allowed engineers to develop tools that could predict the detailed behavior of flow fields and to approximate dynamic fluvial processes, such as channel evolution in response to detailed flow fields. Recent examples of developments using this engineering approach that have potential for ecological application include: Capra et al. (2003), Cioffi and Gallerano (2003), Emery et al. (2003), Sauvage et al. (2003), and Pasternack et al. (2004).

Management needs. Neither the synthetic nor the engineering approaches, by themselves, are adequate to develop science-based management strategies for rivers. At one end of the spectrum of research, advances made by synthesists have been largely conceptual in nature. Their findings are usually presented in a relative sense because the primary goal was to guide the development of research or to develop broad conservation strategies for systems that could not be convincingly reduced to mechanistic models. The synthetic approach is very useful from a heuristic or theoretical standpoint, but cannot be used a priori to address many river management issues because it is insufficiently quantitative at the scale at which management decisions must be made. Relative descriptions of important processes typical of the synthesists cannot be used to support equitable water resources developments when decision-makers expect analysis results as typically provided by incremental, engineering tools. However, the synthetic approach can be used a posteriori to guide large-scale restoration schemes (e.g., Poff et al. 1997) or to set general guidelines for conservation action on individual rivers (e.g., Richter et al. 1997). Implementation of the synthesist approach in the context of site-specific river management is typically viewed as “experimental” (e.g., Stanford et al. 1996, Poff et al. in prep.).

The engineering approaches suffer the inverse problem identified for the synthetic approaches. That is, the engineering approaches are generally restricted to simulating physical or chemical processes that can be approximated by the Eulerian-based approach of discretizing complex geometries with a grid (or mesh) and then applying sets of governing equations to each node. The engineering approach works well for simulating certain processes that are easy to aggregate into control volumes (e.g., water flow or water quality) but appears to work less well to describe the response to processes that are not easy to aggregate (e.g., habitat selection by fish and macroinvertebrates).

The issue of *data* was considered in the section Data needs, but, in addition, as long as the techniques and approaches used by engineers and synthesists remain separate, it is unlikely that river managers will have the *tools* necessary to manage for sustainability or to develop restoration plans for impacted rivers. Therefore, the logical next step is to develop approaches that allow the tools of synthesists and engineers to be coupled together. One way to achieve this objective is illustrated in Figure 14.

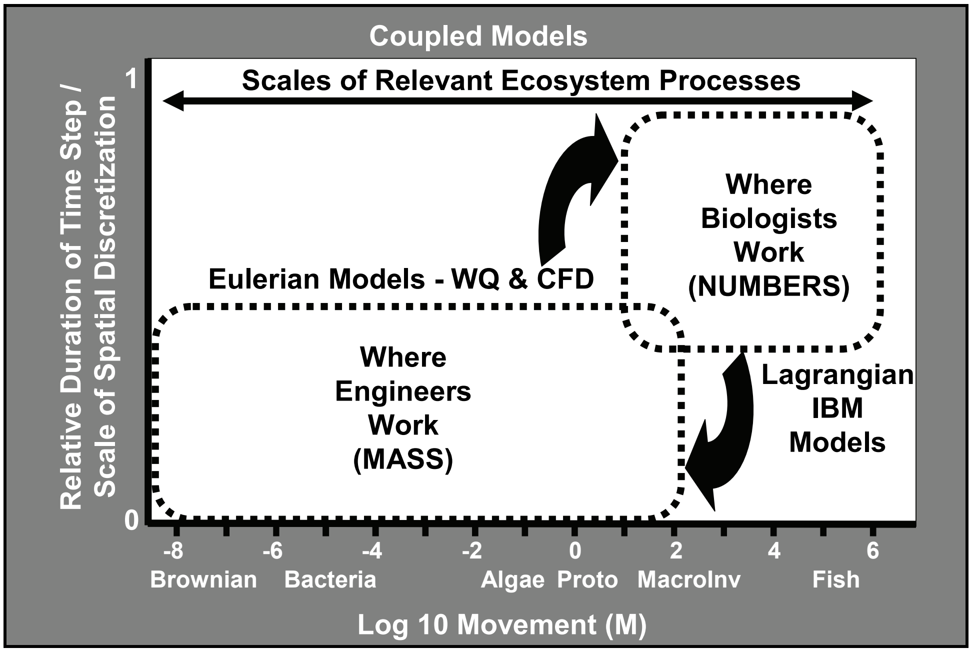


Figure 14. Relationship between temporal and spatial scales with scientific discipline. Note that the approaches used by synthesists and engineers must be coupled to simulate the hierarchy of scales typical of systems such as large rivers. Legend: Brownian=Brownian motion, Proto=protozoans, MacroInv=macroinvertebrates, WQ=water quality modeling, and CFD=computational fluids dynamic modeling

Communication

It is critically important to improve the interchange of information, not only among academics and managers, but also with the interested public. All scientists, regardless of profession, must take an active role in scientific outreach. Academics need to be fully informed about, and be sensitive to, the time and resource constraints that resource mangers have placed upon them. Conversely, academic institutions, in order to ensure their societal relevance, should develop academic review processes that value contributions to the public sector.

User-conflicts must be able to be weighed against the integrity of the ecosystem. Consider the problem of conveying to the general public the advantages of natural variability. Ecologically, we know that extreme events often reset communities and cause episodic functions to occur. However, the average layperson perceives a “healthy system” as one that remains constant. Deflections from the status quo are often alarming to the public and result in an outcry for intervention. A major challenge is to educate society of the fact that natural variability is a prerequisite for sustainability and protection of biodiversity.

The EU Water Framework Directive requires Member States to provide information to, and consult with, stakeholders; Member States are also encouraged to actively engage and involve stakeholders in collaborative activities. Across Europe there is more than two decades of effort, with variable success, requiring scientists to move away from their laboratory bench to engage in knowledge and technology transfer with the range of stakeholders. Today, there is a recognition that internationally competitive science requires specialist expertise, as does interfacing science with stakeholders and the public. It is one thing to encourage a culture within science that acknowledges and even promotes outreach, it is another to expect all experts to be able to participate in outreach effectively. The roles of skilled scientific translators, facilitators, and mediators have now become highly regarded. Experts in such positions serve to help digest and apply academic research to management issues and to promote knowledge transfer to the full spectrum of stakeholders. They also provide research scientists with feedback to identify and prioritize new research challenges and to advance a clearer perspective of critical problems in river basin management.

A Way Ahead

The Workshop included engineers, researchers (ecologists, geomorphologists, and hydrologists) who could be categorized as synthesists, and resource managers. The group concluded that the direction toward improved management of large rivers should include:

a. *Increased communication among basic scientists, applied scientists, and resource managers.* Currently, professional specialization limits adequate communications so that basic researchers are partially isolated from applied scientists. The linkage between applied scientists and resource managers is more robust.

b. *Increased collaboration in tool building.* The three groups must collaborate to build the tools that ultimately will be transferred to river managers as well as feedback on tool performance to the tool builders and researchers. One of the most urgent needs is for indicators of processes and functions.

c. *Increased collaboration between researchers.* As demonstrated by this workshop, improved management of rivers must include international cooperation because (a) integration of knowledge and approaches advanced by different nations with different histories, cultures, and traditions offers exciting opportunities for accelerating progress and (b) there are a finite number of large rivers in the world and progress on sustainable management is lagging advances in natural resource exploitation.

d. *Development of a consensus, long-term research vision.* Scientists and managers working on large rivers must agree on a common vision that will guide research, tool development, and management concepts. There would be considerable benefits if such a long-term vision could incorporate international collaboration. One element of this long-term vision must be the development of models to evaluate the complex effects of abiotic variations upon habitats, species and communities, and their interactions.

e. *Move to establish naturalistic models.* These are management models that incorporate the range of special habitats that are typical of natural rivers but lost to most regulated systems such as springs, ponds, sediment bars, woody-debris accumulations, and wooded islands and riparian zones.

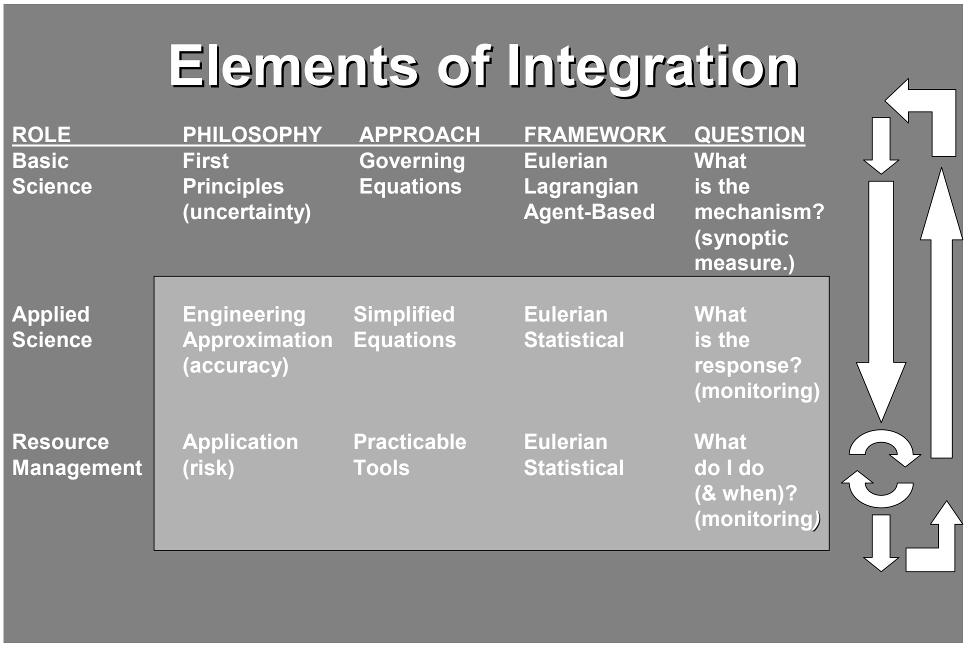
f. *Drive to understand variability.* There is an urgent need to determine the variability of key abiotic parameters over the range of spatial scales, to measure and model the effects of these variations upon biota, habitats, and ecosystems, to understand the time scales and mechanisms of ecosystem response to hydrological change, and to educate the public about the importance of variability in sustaining riverine ecosystems.

g. *Commitment to long-term research and monitoring.* The ultimate approach for improved management of large rivers is commitment to the long-term research designed to better describe processes that characterize large rivers. Coupled data designed to address abiotic–biotic responses over (at least) decadal time scales are required to underpin the progressive advancememnt of key models. Again, international collaboration would be beneficial to encompass different types of river and different socioeconomic settings.

h. *Commitment to advancing knowledge-based approaches to management.* This requires more and different data and integrative frameworks that embrace the traditions of both synthesists and holists. Institutional mechanisms (adaptive management) must be introduced to ensure that knowledge gained through science and project monitoring can be incorporated into management action.

The basic science of both engineers and synthesists must be coupled together at a “first principles” level in an hypothesis-testing setting to develop defensible applied scientific tools (Figure 15). Applied science tools are used by resource managers to develop management plans for sustainable development of large rivers. Adaptive management (represented by the arrows in Figure 15) allows a constant feedback between the three integrating levels, particularly between the tool builders and end-users. Clearly, we also need indicators of management performance to advance adaptive management.

Across Europe, a key political driver to advance research excellence is to increasingly intensify competition among researchers and research groups. Some view the primary aim of the proposed European Research Council as fostering greater competition between already internationally competitive research groups. There is no doubt that competition does result in maintaining and advancing quality. This Workshop and its primary recommendations focus, however, on collaboration between internationally competitive research groups as the more efficient route to accelerating and directing the advancement of knowledge on river ecosystems so as to realize the ambition of achieving the integration of human and environmental water needs in river basin management.



Elements of Integration

ROLE	PHILOSOPHY	APPROACH	FRAMEWORK	QUESTION
Basic Science	First Principles (uncertainty)	Governing Equations	Eulerian Lagrangian Agent-Based	What is the mechanism? (synoptic measure.)
Applied Science	Engineering Approximation (accuracy)	Simplified Equations	Eulerian Statistical	What is the response? (monitoring)
Resource Management	Application (risk)	Practicable Tools	Eulerian Statistical	What do I do (& when)? (monitoring)

Figure 15. Relationship between the three sequential levels of integration necessary to develop good resource management tools

The Workshop proposed the creation of a Linked European–American Research Network on Rivers (LEARNER) to facilitate collaboration between applied scientists working on larger rivers. No single large river can be studied to the extent necessary to fully describe catchment, riparian, groundwater, and in-channel processes that, integrated together, manifest themselves as the state of the river because of the challenges outlined above. International collaboration will take advantage of the scientific economies of scale that occur when researchers working on larger river systems are able to share data and their understanding of river processes.

References

Abernethy, V. J., and Willby, N. J. (1999). "Changes along a disturbance gradient in the density and composition of propagule banks in floodplain aquatic habitats," *Plant Ecology* 140, 177–190.

Amoros, C., and Petts, G. E. (ed.) (1993). *Hydrosystèmes Fluviaux*, Masson, Paris.

Andersen, J. (1968). "The effect of inundation and choice of hibernation sites of Coleoptera living on river banks," *Norsk Entomologisk Tidsskrift* 15, 115–133.

____________. (1969). "Habitat choice and life history of *Bembidion* (Col., Carabidae) on river banks in central and northern Norway," *Norsk Entomologisk Tidsskrift* 17, 17–65.

____________. (1978). "The influence of substratum on the habitat selection of Bembidiini (Col., Carabidae)," *Norwegian Journal of Entomology* 25, 119–138.

____________. (1979). "Interspecific relations within the genus *Bembidion* (Col., Carabidae)," *Entomologisk Tidskrift* 100, 151–155.

____________.. (1983a). "The habitat distribution of species of the tribe Bemdidiini (Coleoptera, Carabidae) on banks and shores in northern Norway," *Notulae Entomologicae* 63, 131–142.

____________. (1983b). "The life cycles of the riparian species of *Coleoptera*, Carabidae) in northern Norway," *Notulae Entomologicae* 63, 195–202.

____________. (1985a). "Ecomorphological adaptations of riparian Bembidiini species (Coleoptera, Carabidae)," *Entomologica Generalis* 12, 41–46.

____________. (1985b). "Low thermo-kinesis, a key mechanism in habitat selection by riparian *Bembidion* (Carabidae) species," *Oikos* 44, 499–505.

____________. (1988). "Resource partitioning and interspecific interactions among riparian *Bembidion* species (Coleoptera: Carabidae)," *Entomologica Generalis* 13, 47–60.

Andersson E., Nilsson, C., and Johansson, M. E. (2000a). "Effects of river fragmentation on plant dispersal and riparian flora," *Regulated Rivers* 16, 83–89.

Andersson E., Nilsson, C., and Johansson, M. E. (2000b). “Plant dispersal along boreal rivers and its relation to the riparian flora,” *Journal of Biogeography* 27, 1095–1106.

Angradi, T.R. (1994). “Trophic linkages in the lower Colorado River—multiple stable isotope evidence,” *Journal of the North American Benthological Society* 13, 479–495.

Anon. (1995). *Biodiversity*: *The UK Steering Group Report*, Department of the Environment, London.

Archer, D., and Newson, M. (2002). “The use of indices of flow variability in assessing the hydrological and instream habitat impacts of upland forestation and drainage,” *Journal of Hydrology* 268, 244–258.

Armstrong, J. D., Holm, C. F., and Gilvear, D. J. (2002). “Holding station against the flow: A reply to Ibbotson and Dunbar,” *Journal of Fish Biology* 61, 308–312.

Arscott, D. B., Tockner, K., and Ward, J.V. (2000). “Aquatic habitat structure and diversity along the corridor of an Alpine floodplain river (The Fiume Tagliamento),” *Archiv für Hydrobiologie* 149, 679–704.

Arscott, D. B., Tockner, K., van der Nat, D., and Ward, J. V. (2002). “Aquatic habitat dynamics along a braided alpine river ecosystem (Tagliamento River, Northeast Italy),” *Ecosystems* 5, 802–814.

Arthington, A. H. (1998). *Comparative evaluation of environmental flow techniques: Review of holistic methodologies,* Land and Water Resources, Research and Development Corporation*,* Occasional Paper Series 26/98, Canberra, Australia.

Arthington, A. H., and Zalucki, J. M. (ed.) (1998). *Comparative evaluation of environmental flow assessment techniques: Review of methods*, Land and Water Resources, Research and Development Corporation, Occasional Paper No. 27/98, Canberra, Australia.

Arthington, A. H., Brizga, S. O., and Kennard, M. J. (1998). *Comparative evaluation of environmental flow assessment techniques: Best practice framework.* Land and Water Resources Research and Development Corporation Occasional Paper No. 25/98, Canberra, Australia.

Baker, A. (2001). “Fluorescence excitation-emission matrix characterisation of some sewage impacted rivers,” *Environmental Science and Technology* 35, 948–953.

Baker, A. (2002). “Fluorescence properties of some farm wastes: Implications for water quality monitoring,” *Water Research* 36, 189–194.

Baptist, M. J., and Mosselman, E. (2002). “Biogeomorphological modelling of secondary channels in the Waal river.” *River Flow 2002 Symposium*, Louvain-la-Neuve, Belgium, D. Bousmar and Y. Zech, ed., 773–782.

Baptist, M. J., Penning, W. E., Duel, H., Smits, A. J. M., Geerling, G. W., Van der Lee, G. E. M., and Van Alphen, J. S. L. (2004). “Assessment of cyclic floodplain rejuvenation on flood levels and biodiversity in the Rhine River,” *River Research and Applications*, special issue IRMA-projects, 20, 285–298.

Bastow, J. L., Sabo, J. L., Finlay, J. C., and Power, M. E. (2002). "A basal aquatic-terrestrial trophic link in rivers: Algal subsidies via shore-dwelling grasshoppers," *Oecologia* 131, 261–268.

Bates, A. J., Sadler, J. P., Fowles, A. P., and Butcher, C. R. (in prep.). "Spatial dynamics of beetles living on exposed riverine sediments in the Upper River Severn: Method development and preliminary results," *Aquatic conservation—Marine and Freshwater Ecosystems.*

Bayley, P. B. (2002). "A review of studies on responses of salmon and trout to habitat change, with potential for application in the Pacific Northwest," Report to the Washington State Independent Science Panel, Department of Fisheries and Wildlife, Oregon State University.

Bartholow, J. M., Laake, J. L., Stalnaker, C. B., and Williamson, S. C. (1993). "A salmonid population model with emphasis on habitat limitations," *Rivers* 4, 265–279.

Baumgärter, R., and Grüneklee, W. (2002). "Sukzession nach Dammbruch auf ehemaligen Ackerflächen in der Rheinaue: Ergebnisse nach 17 Jahren ungestörter Sukzession auf der Rheininsel Kühkopf," *Natur and Landschaft* 77, 269–273.

Bedford, B. L. (1996). "The need to define hydrologic equivalence at the landscape scale for freshwater wetland mitigation," *Ecological Applications* 6, 57–68.

Bell, D., Petts, G. E., and Sadler, J. P. (1999). "The distribution of species in the wooded riparian zone of three rivers in western Europe," *Regulated Rivers* 15, 141–158.

Bendix, J., and Hupp, C. R. (2000). "Hydrological and geomorphological impacts on riparian plant communities." *Hydrological Processes* 14, 2977–2990.

Brinson, M. M., and Rheinhardt, R. (1996). "The role of reference wetlands in functional assessment and mitigation," *Ecological Applications* 6, 69–76.

Bonn, A. (2000). "Flight activity of carabid beetles on a river margin in relation to fluctuating water levels, *Natural History and Applied Ecology of Carabid Beetles*, Brandmayr, P., Lövei, G. L., Brandmayr, T. Z., Casale, A. and A. V. Taglianti, ed., Pensoft, Sofia, 147–160.

Bonn, A., Hagen, K., and Wohlgemuth-Von Reiche, D. (2002). "The significance of flood regimes for carabid beetle and spider communities in riparian habitats—A comparison of three major rivers in Germany," *River Research and Applications* 18, 43–64.

Bonn, A., and Kleinwächter, M. (1999). "Microhabitat distribution of spider and ground beetle assemblages (Araneae, Carabidae) on frequently inundated river banks of the River Elbe," *Zeitschrift für Ökologie und Naturschutz* 8, 109–123.

Botosaneanu, L. (1998). "Sources: aux portes du styx," *Studies in Crenobiology: The biology of springs and springbrooks*, Botosaneanu, L., ed., Backhuys, Leiden, 229–250.

Boulton, A. J. (1989). “Over-summering refuges of aquatic macroinvertebrates in two intermittent streams in Central Victoria,” *Transactions of the Royal Society of South Australia* 113, 23–34.

____________. (1993). “Stream ecology and surface-hyporheic exchange : Implications, techniques and limitations,” *Freshwater Resources* 44, 553-564.

Boulton, A. J., and Lake, P. S. (1992). “The ecology of two intermittent streams in Victoria, Australia. III. Temporal changes in faunal composition,” *Freshwater Biology* 27, 123–138.

Bovee, K. D. (1982). “A guide to stream habitat analysis using the Instream Flow Incremental Methodology,” FWS/OBS 82/26, U.S. Fish and Wildlife Service, Fort Collins, Colorado.

Brewer, P. A., and Lewin, J. (1998). “Planform cyclicity in unstable reach: Complex fluvial response to environmental change,” *Earth Surface Processes and Landforms* 23, 989–1008.

Breil, P. (1997). “Regional hydrological determinants for instream ecology,” Flow Regimes from International Experimental and Network Data, third quadriennal report : 1994–1997, International Programme of Hydrology, UNESCO, Phases IV and V, Cemagref Edition, 386–390.

Brittain, J. E., and Nilsson, C. (ed.) (1996). “Remedial strategies in regulated rivers,” *Regulated Rivers, Special Issue* 12, 345–561.

Brooks, A. P., and Brierley, G. J. (2002). “Mediated equilibrium: the influence of riparian vegetation and wood on the long-term evolution and behaviour of a near-pristine river,” *Earth Surface Processes and Landforms* 27, 343–367.

Brooks, C. P. (2003). “A scalar analysis of landscape connectivity,” *Oikos* 102, 433–439.

Brown, A. G. (1997). “Biogeomorphology and diversity in multiple-channel river systems,” *Global Ecology and Biogeography Letters* 6, 179–185.

Brunke, M., Hofann, A., and Pusch, M. (2001). “Use of mesohabitat-specific relationships between flow velocity and river discharge to assess invertebrate minimum flow requirements,” *Regulated Rivers* 17, 667–676.

Bunn, A. G., Urban, D. L., and Keitt, T. H. (2000). “Landscape connectivity: a conservation application of graph theory,” *Journal of Environmental Management* 59, 265–278.

Bunn S. E., and Arthington, A. H. (2002). “Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity,” *Environmental Management* 30, 455–467 and 492–507.

Bunn, S. E., Davies, P. M., and Winning, M. (2003). “Sources of organic carbon supporting the food web of an arid zone floodplain river*,” Freshwater Biology* 48, 619–635.

Burbrink, F. T., Phillips, C. A. and Heske, E. J. (1998). “A riparian zone in southern Illinois as a potential dispersal corridor for reptiles and amphibians,” *Biological Conservation* 86, 107–115.

Capra, H., Sabaton, C., Gournaud, V., Souchon, Y., and Lim, P. (2003). "A population dynamics model and habitat simulation as a tool to predict brown trout demography in natural and bypassed stream reaches," *River Research and Applications* 19, 551–568.

Capra, H., Breil, P., and Souchon, Y. (1995). "A new tool to interpret magnitude and duration of fish habitat variations," *Regulated Rivers* 10, 291–289.

Cardwell, H. E., Jager, H. I., and Sale, M. J. (1996). "Designing instream flows to satisfy fish and human water needs," *Journal of Water Resources Planning and Management—ASCE*, 122, 356–363.

Castleberry, D. T., Cech, Jr., J. J., Erman, D. C., Hankin, D., Healey, M., Kondolf, G. M., Mangel, M., Mohr, M., Moyle, P. B., Nielsen, J., Speed, T. P., and Williams, J. G. (1996). "Uncertainty and instream flow standards," *Fisheries* 21, 20–21.

Cattanéo, F., Carrel, G., Lamouroux, N., and Breil, P. (2001). "Relationship between hydrology and cyprinid reproductive success in the lower Rhône at Montélimar, France." *Archiv für Hydrobiologie* 151(3), 427–450.

Cattanéo, F., Lamouroux, N., Breil, P., and Capra, H. (2002). "The influence of hydrological and biotic processes on brown trout (*Salmo trutta*) population dynamics." *Canadian Journal of Fisheries and Aquatic Sciences* 59, 12–22.

Charles, S., Bravo de la Parra, R., Mallet, J. P., Persat, H., and Auger, P. (1998). "Population dynamics modelling in an hierarchical arborescent river network: an attempt with *Salmo trutta*," *Acta Biotheoretica* 46, 223–234.

Chaumot, A., Charles, S., Auger, P., Flammarion, P., and Garric, J. (1999). "Development of an ecotoxicological model in an arborescent river network: An attempt with brown trout population," *Aspects of Applied Biology* 53, 131–136.

Church, M. (2002). "Geomorphic thresholds in riverine landscapes," *Freshwater Biology* 47, 541–558.

Cioffi, F., and Gallerano, F. (2003). "A 2D self-adaptive hydrodynamic scheme for the assessment of the effects of structures on flooding phenomena in river basins." *River Research and Applications* 19, 1–26.

Clausen, B., and Biggs, B. J. F. (1997). "Relationships between benthic biota and hydrological indices in New Zealand streams," *Freshwater Biology* 38, 327–342.

Clausen, B., and Biggs, B. J. F. (2000). "Flow variables for ecological studies in temperate streams grouping based on covariance," *Journal of hydrology* 237, 184–197.

Collier, K. J., Bury, S., and Gibbs, M. (2002). "A stable isotope study of linkages between stream and terrestrial food webs through spider predation," *Freshwater Biology* 47, 1651–1659.

Colwell, R. K. (1974). "Predictability, constancy, and contingency of periodic phenomena," *Ecology* 55, 1148–1153.

Connell, J. H. (1978). “Diversity in tropical rainforests and coral reefs,” *Science* 199, 1302–1310.

Connell, J. H., and Wayne, P. S. (1983). “On the evidence needed to judge ecological stability or persistence,” *The American Naturalist* 121, 798–824.

Covitch, A. P., Crowl, T. A., and Scatena, F. N. (2003). “Effects of extreme low flows on freshwater shrimps in a perennial tropical stream,” *Freshwater Biology* 48, 1199–1206.

Darveau, M., Labbe, P., Beauchesne, P., Belanger, L., and Huot, J. (2001). “The use of riparian forest strips by small mammals in a boreal balsam fir forest,” *Forest Ecology and Management* 143(1–3), 95–104.

Daufresne, M., Capra, H., and Lamouroux, N. (2003). “Long-term changes within the invertebrate and fish communities of the Upper Rhône River: Effects of climatic factors,” *Global Change Biology* 10, 124–140.

DeAngelis, D. L., Rose, K. A., and Huston, M. A. (1994). “Individual-oriented approaches to modeling ecological populations and communities,” *Frontiers in Mathematical Biology,* Levin, S. A., ed,. Springer, Berlin, 390–410.

De Waal, L. C., Child, L. E., Wade, P. M., and Brock, J. H. (1994). *Ecology management of invasive riverside plants*, Wiley and Sons, Chichester.

Decamps, H. (1996). “The renewal of floodplain forests along rivers: A landscape perspective,” *Internationale Vereinigung Fur Theoretische Und Angewandte Limnologie* 26, 35–59.

De Leo, G. A., and Levin, S. (1997). “The multifaceted aspects of ecosystem integrity,” *Conservation Ecology* 1, 3–24.

Dent, C. L., Grimm, N. B., and S. G. Fisher (2001). “Multiscale effects of surface-subsurface exchange on stream water nutrient concentrations,” *Journal of the North American Benthological Society*. 20(2), 162–181.

Desender, K. (1989). “Ecomorphological adaptations of riparian carabid beetles,” *Proceedings of the symposium 'Invertebrates of Belgium,'* L'Institut Royal des Science Naturelles de Belgique, Brussels, 309–314

Dister, E. (1980). “Geobotanische Untersuchungen in der Hessischen Rheinaue als Grundlage für die Naturschutzarbeit,” Ph.D. diss.

Dolloff, C. A., Flebbe, P. A., and Owen, M. D. (1994). “Fish habitat and fish populations in a southern appalachian watershed before and after hurricane Hugo,” *Transactions of the American Fisheries Society* 123, 668–678.

Drake, M. (2001). “The importance of temporary waters for Diptera (true flies),” *Freshwater Forum* 17, 26–39.

Dunbar, M. J., Gustard, A., Acreman, M. C., and Elliott, C.R.N. (1998). “Overseas approaches to setting River Flow Objectives,” Institute of Hydrology, U.K., R & D Technical report W6-161.

Dynesius M., and Nilsson, C. (1994). “Fragmentation and flow regulation of river systems in the northern third of the world,” *Science* 266, 753–762.

Edwards, P. J., Kollmann, J., Gurnell, A. M., Petts, G. E., Tockner, K., and Ward, J.V. (1999). “A conceptual model of vegetation dynamics on gravel bars of a large Alpine river,” *Wetlands Ecology and Management* 7, 141–153.

Elliott J. M. (1994). *Quantitative Ecology and the Brown Trout,* Oxford University press, Oxford, UK.

Emery, J. C., Gurnell, A. M., Clifford, N. J., Petts, G. E., Morrissey, I. P., and Soar, P. J. (2003). “Classifying the hydraulic performance of riffle-pool bedforms for habitat assessment and river rehabilitation design,” *River Research and Applications* 19, 533–550.

Environment Agency. (2001) “Water resources for the future: A strategy for England and Wales,” Environment Agency, Bristol.

Erman, N. A. (1998). “Invertebrate richness and Trichoptera phenology in Sierra Nevada (California, USA) cold springs: Sources and variation,” *Studies in Crenobiology: The biology of springs and springbrooks,* Botosaneanu, L., ed., Backhuys, Leiden, 95–108.

Erman, N. A., and D. C. Erman (1995). “Spring permanence: Trichoptera species richness, and the role of drought,” *Journal of the Kansas Entomological Society*, 68(2), 50–64.

Essington, T. E. (2003). “Development and sensitivity analysis of bioenergetics models for skipjack tuna and albacore: a comparison of alternative life histories,” *Transactions of the American Fisheries Society* 132, 759–770.

Estes, C. C. (1996). “Annual summary of instream flow reservations and protection in Alaska,” Alaska Department of Fish and Game, Fisheries Data Series No. 96-45, Anchorage.

Eyre, M. D., Lott, D. A., and Luff, M. L. (2001a). “The rove beetles (*Coleoptera, Staphylinidae*) of exposed riverine sediments in Scotland and northern England: Habitat classification and conservation aspects,” *Journal of Insect Conservation* 5, 173–186.

Eyre, M. D., Luff, M. L., and Phillips, D. A. (2001b). “The ground beetles (Coleoptera : Carabidae) of exposed riverine sediments in Scotland and northern England,” *Biodiversity and Conservation* 10, 403–426.

Evans, E. C., and Petts, G. E. (1997). “Hyporheic temperature patterns within riffles,” *Hydrological Sciences Journal* 42(2), 199–213.

Fausch, K. D., Hawkes, C. L., and Parsons, M. G. (1988). “Models that predict standing crop of stream fish from habitat variables: 1950–85,” General Technical Report PNW-GTR-213*,* U.S. Department of Agriculture, Forest Service Pacific Northwest Research Station, Portland, Oregon.

Fellows, C. S., Valett, M. H., and Dahm, C. N. (2001). “Whole-stream metabolism in two montane streams: Contribution of the hyporheic zone.” *Limnological Oceanography* 46(3), 523–531.

Feminella, J. W. (1996). “Comparisons of benthic macroinvertebrate assemblages in small streams along a gradient of flow permanence,” *Journal of the North American Benthological Society* 15, 651–669.

Findlay, S. E. G., and Sinsabaugh, R. L. (2003). *Aquatic Ecosystems: Interactivity of DOM*. Academic Press, New York.

Fischer-Antze, T., Stoesser, T., Bates, P., and Olsen, N. R. B. (2001). "3D numerical modelling of open-channel flow with submerged vegetation," *Journal of Hydraulic Research* 39, 303–310.

Fischer, H., Sukhodolov, A., Wilczek, S., and Engelhardt, C. (2003). "Effects of flow dynamics and sediment movement on microbial activity in a lowland river," *River Research and Applications* 19, 473–482.

Fleishman, E., Ray, C., Sjogren-Gulve, P., Boggs, C. L., and Murphy, D. D. (2002). "Assessing the roles of patch quality, area, and isolation in predicting metapopulation dynamics," *Conservation Biology* 16, 706–716.

Fortin, M.-J., Boots, B., Csillag, F., and Remmel, T. K. (2003). "On the role of spatial stochastic models in understanding landscape indices in ecology," *Oikos* 102, 203–212.

Fowles, A. P. (1989). "The Coleoptera of shingle banks on the River Ystwyth, Dyfed," *Entomologist's Record* 101, 209–221.

Framenau, V., Dieterich, M., Reich, M., and Plachter, H. (1996a). "Life cycle, habitat selection and home ranges of *Arctosa cinerea* (Fabricius, 1771) (Araneae: Lycosidae) in a braided section of the Upper Isar (Germany, Bavaria)," *B Revue Suisse de Zoologie Rev*. 223–234

____________. (1996b). "Migration and prey of *Arctosa cinerea* (Fabricius, 1771) on a braided section of an Alpine River," *Verhandlungen der Gesellschaft für Ökologie* 26, 369–376.

Framenau, V. W., Manderbach R., and Baehr, M. (2002). "Riparian gravel banks of upland and lowland rivers in Victoria (south-east Australia): arthropod community structure and life-history patterns along a longitudinal gradient," *Australian Journal of Zoology* 50, 103–123.

Gaston, K. J. (1994). *Rarity*. Chapman & Hall, London.

Gibbins, C. N., Dilks, C. F., Malcolm, R., Soulsby, C., and Juggins, S. (2001). "Relationships between invertebrates and hydrological variability in Cairngorm mountain streams," *Hydrobiologia* 462, 205–219.

Gibbins, C. N., Moir, H., Webb, J., and Soulsby, C. (2002). "Assessing discharge use by spawning Atlantic salmon: A comparison of discharge electivity indices and PHABSIM simulations," *Rivers Research and Application* 18, 383–395.

Ginot, V., Le Page, C., and Souissi, S. (2002). "A multi-agents architecture to enhance end-user individual based modelling," *Ecological Modelling* 157, 23–41.

Gold, A. J., and Kellogg. D. Q. (1997). "Modelling internal processes of riparian buffer zones," *Buffer Zones: Their Processes and Potential in Water Protection*, N. Haycock, T. Burt, K. Goulding and G. Pinay, ed., Harpenden, UK: Quest Environmental, 192–207.

Gooch, J. L., and D. S. Glazier (1991). "Temporal and spatial patterns in mid-Applachian springs," *Memoirs of the Entomological Society of Canada* 115, 29–49.

Goodson, J. M., Gurnell, A. M., Angold, P. G., and Morrissey, I. P. (2001). "Riparian seed banks: structure, process and implications for riparian management," *Progress in Physical Geography* 25(3), 301–325.

____________. (2002). "Riparian seed banks along the lower River Dove, UK: Their structure and ecological implications." *Geomorphology* 47(1), 45–60.

Gore, J. A., and Mead, J. (2000). "The benefits and dangers of ecohydrological models to water resource management decisions," *Ecohydrology: A New Paradigm*, United Nations/UNESCO, Geneva and Cambridge University Press.

Gouraud, V., Baglinière, J. L., Baran, P., Sabaton, C., Lim, P., and Ombredane, D. (2001). "Factors regulating brown trout populations in two French rivers: Application of a dynamic population model," *Regulated Rivers* 17, 557–569.

Gouraud V., Baran, P., Lim, P., and Sabaton, C. (1999). "Dynamics of a population of brown trout (*Salmo trutta*) and fluctuations in physical habitat conditions—experiments on a stream in the Pyrenees; first results," *Rivers Fisheries, Fishing News Books,* Cowx, I.G., ed., Blackwell Science, Oxford, 126–142.

Gran, K., and Paola, C. (2001). "Riparian vegetation controls on braided stream dynamics," *Water Resources Research* 37, 3275–3283.

Gregory, S. V., Swanson, F. J., McKee, W. A., and Cummins, K.W. (1991). "An ecosystem perspective of riparian zones," *Bioscience* 41(8), 540–551.

Grimm, V. (1999). "Ten years of individual-based modelling in ecology: What have we learned and what could we learn in the future?" *Ecological Modelling* 115, 129–148.

Gunn, J., Hardwick, P., and Wood, P. J. (2000). "The invertebrate community of the Peak-Speedwell cave system, Derbyshire, England—pressures and considerations for conservation management," *Aquatic Conservation: Marine and Freshwater Ecosystems* 10, 353–369.

Gurnell, A. M., and Petts, G. E. (2002). "Island-dominated landscapes of large floodplain rivers: A European perspective," *Freshwater Biology* 47, 581–600.

Hammond, P. M. (1998). "Riparian and floodplain arthropod assesmblages: Their characteristics and rapid assessment," *United Kingdom floodplains,* R. G. Bailey, P. V. Jos, and B. R. Sherwod, ed., Westbury Press, Otley, 237–282.

Hannah, D. M., Malcolm, I. A., Soulsby, C., and Youngson, A. F. (in press). "Heat exchange and temperature behaviour in a salmon spawning stream in the Cairngorms, Scotland: Seasonal and subseasonal dynamics," *Rivers Research and Applications* 20.

Harris, N. M., Gurnell, A., Hannah, D. M., and Petts, G. E. (2000). "Classification of river regimes: A context for hydroecology," *Hydrological Processess* 14, 2831–2848.

Harris, D. D., Hubert, W. A., and Wesche, T. A. (1991). "Brown Trout population and habitat response to enhanced minimum flow in Douglas Creek, Wyoming," *Rivers* 2, 285–294.

Harrison, S. (1991). "Local extinction in a metapopulation context—An empirical evaluation," *Biological Journal of the Linnean Society* 42, 73–88.

Harner, M. J., and Stanford, J. A. (2003). "Differences in cottonwood growth between a losing and a gaining reach of an alluvial floodplain," *Ecology* 84, 1453–1458.

Harpman, D. A., Sparling, E. W., and Waddle, T. J. (1993). "A methodology for quantifying and valuing the impacts of flow changes on a fishery," *Water Resources Research* 29, 575–582.

Haycock, N., Burt, G. T., and Pinay, G. (ed.) (1997). *Buffer Zones: Their Processes and Potential in Water Protection*, Harpenden, U.K., Quest Environmental.

Hayes, J. W., Stark, J. D., and Shearer, K. A. (2000). "Development and test of a whole-lifetime foraging and bioenergetics growth model for drift-feeding brown trout," *Transactions of the American Fisheries Society* 129, 315–322.

Hayford, H., and Herrmann, S. J. (1998). "Migration patterns of four macroinvertebrates along a rheocrene thermal spring," *Studies in Crenobiology: The biology of springs and springbrooks,* Botosaneanu, L., ed., Backhuys, Leiden, 75–83.

Henschel, J. R., Mahsberg, D., and Stumpf, H. (2001). "Allochthonous aquatic insects increase predation and decrease herbivory in river shore food webs," *Oikos* 93, 429–438.

Hering, D. (1998). "Riparian beetles (Coleoptera) along a small stream in the Oregon Coast Range and their interactions with the aquatic environment," *Coleopterists Bulletin* 52, 161–170.

Hering, D., and Plachter, H. (1997). "Riparian ground beetles (Coleoptera, Carabidae) preying on aquatic invertebrates: A feeding strategy in alpine floodplains," *Oecologia* 111, 261–270.

Hilderbrand, R. H. (2003). "The roles of carrying capacity, immigration, and population synchrony on persistence of stream-resident cutthroat trout," *Biological Conservation* 110, 257–266.

Hildrew, A. G., and Townsend, C. R. (1987). "Organization in freshwater benthic communities," *Organization of Communities, Proc 27th Symposium British Ecological Society*, Gee, J. H. R., Giller, P. S., ed., Blackwell Scientific, Oxford, 347–371.

Hoffsten, P.-O., and Malmqvist, B. (2000). "The macroinvertebrate fauna and hydrogeology of springs in central Sweden," *Hydrobiologia* 436, 91–104.

Holland, M. M. (1988). “SCOPE/MAB technical consultations on landscape boundaries,” *Biology International*, Special Issue 17, 47–106.

Holland, M. M., Whigham, D., and Gopal, B. (1990). “The characteristics of Wetland Ecotones,” *Land/Inland Water Ecotones,* Naiman, R. J., and H. Decamps ed., UNESCO's Man and the Biosphere Book Series, 4, The Parthenon Publishing Group, London and New York, 171–198.

Hughes, F. M. R. (1997). “Floodplain biogeomorphology,” *Progress in Physical Geography* 21, 501–529.

Hughes, F. M. R., Adams, W. M., Muller, E., Nilsson, C., Richards, K. S., Barsoum, N., Decamps, H., Foussadier, R., Girel, J., Guilloy, H., Hayes, A., Johansson, M., Lambs, L., Pautou, G., Peiry, J.-L., Perrow, M., Vautier, F., and Winfield, M. (2001). “The importance of different scale processes for the restoration of floodplain woodlands,” *Regulated Rivers* 17, 325–345.

Hughes, N. F. (2000). “Testing the ability of habitat selection theory to predict interannual movement patterns of a drift-feeding salmonid,” *Ecology of Freshwater Fish* 9, 4–8.

Ibbotson, A., and Dunbar, M. (2002). “Are differences between discharge-specific preference functions relevant for the application of PHABSIM?” *Journal of Fish Biology* 61, 305–307.

IUCN. (2003). “Flow. The essentials of environmental flows,” M. Dyson, G. Bergkamp, and J. Scanlon, ed. International Union for Conservation of Nature and Natural Resources, Gland, Switzerland.

Jager, H. I. (2001). “Individual variation in life history characteristics can influence extinction risk,” *Ecological Modelling* 144, 61–76.

Jager, H. I., and Rose, K. A. (2003). “Designing optimal flow patterns for fall Chinook salmon in a Central Valley, California, River,” *North American Journal of Fisheries Management* 23, 1–21.

Jager, H. I., Van Winkle, W., and Holcomb, B. D. (1999). “Would hydrologic climate changes in Sierra Nevada streams influence trout persistence?” *Transactions of the American Fisheries Society* 128, 222–240.

Jager, H. I., Cardwell, H. E., Sale, M. J., Bevelhimer, M. S., Coutant, C. C., and Van Winkle, W. (1997). “Modelling the linkages between flow management and salmon recruitment in rivers,” *Ecological Modelling* 103, 171–191.

Johnston, C. A., and Naiman, R. J. (1987). “Boundary dynamics at the aquatic-terrestrial interface: The influence of beaver and geomorphology,” *Landscape Ecology* 1, 47–57.

Junk, W. J., Bayley, P. B., and Sparks, R. E. (1989). “The flood pulse concept in river-floodplain systems,” *Proc. International Large River Symposium,* D. P. Dodge, ed., Canadian Special Publication Fisheries Aquatic Sciences 106, 110–127.

Kaiser, H. (1979). “The dynamics of populations as result of the properties of individual animals,” *Fortschr. Zool.* 25, 109–136.

Kaiser et al. (2003). “Solid-state and multidimensional solution-state NMR of solid phase extracted and ultrafiltered riverine dissolved organic matter,” *Environmental Science and Technology* 37, 2929–2935.

Karaus, U., Alder, L., and Tockner, K. (2005). “Concave islands: Diversity and dynamics of parafluvial ponds in a gravel-bed river,” *Wetlands* 25, 26-37.

Karr, J. R. (1999). “Defining and measuring river health,” *Freshwater Biology* 41, 221–234.

Kawaguchi, Y., and Nakano, S. (2001). “Contribution of terrestrial invertebrates to the annual resource budget for salmonids in forest and grassland reaches of a headwater stream,” *Freshwater Biology* 46, 303–316.

Kerr, J. T., and Ostrovsky, M. (2003). “From space to species: Ecological applications for remote sensing,” *Trends in Ecology and Evolution* 18, 299–305.

Kilgo, J. C., Sargent R. A., Chapman B. R., and Miller, K. V. (1998). “Effect of stand width and adjacent habitat on breeding bird communities in bottomland hardwoods,” *Journal of Wildlife Management* 62(1), 72–83.

King, J. M., Tharme, R. E., and De Villiers, M. (ed.) (2000). “Environmental flow assessments for rivers: manual for the Building Block Methodology,” Water Research Commission Technology Transfer Report No. TT131/00, Water Research Commission, Pretoria.

Knopf, F. L., and Samson, F. B. (1994). “Scale perspectives on avian diversity in western riparian ecosystems,” *Conservation Biology* 8, 669–676.

Kondolf, G. M., Larsen, E. W., and Williams, J. G (2000). “Measuring and modeling the hydraulic environment for assessing instream flows,” *North American Journal of Fisheries Management* 20, 1016–1028.

Lake, P. S. (2000). “Disturbance, patchiness, and diversity in streams,” *Journal of the North American Benthological Society* 19, 573–592.

Lamouroux, N., and Capra, H. (2002). “Simple predictions of instream habitat model outputs for target fish populations,” *Freshwater Biology* 47, 1543–1556.

Lamouroux, N., and Souchon, Y. (2002). “Simple predictions of instream habitat model outputs for fish habitat guilds in large streams,” *Freshwater Biology* 47, 1531–1542.

Lamouroux, N., Capra, H., Pouilly, M., and Souchon, Y. (1999). “Fish habitat preferences in large streams of southern France,” *Freshwater Biology* 42, 673–687.

Lamouroux, N., Poff, L. N., and Angermeier, P. L. (2002). “Intercontinental convergence of stream fish community traits along geomorphic and hydraulic gradients,” *Ecology* 83, 1792–1807.

Langan, S. J., Wade, A. J., Smart, R., Edwards, A. C., Soulsby, C., Billett, M., Jarvie, H., Cresser, M. S., Owen, R., and Ferrier, R. C. (1997). “The prediction and management of water quality in a relatively unpolluted major

Scottish catchment: current issues and experimental approaches," *Science of the Total Environment* 194/5, 419–435.

Langhans, S., and Tockner, K. (in prep.). "Floating organic matter: A functional link along river corridors."

Latterell, J. J., Fausch, K. D., Gowan, C., and Riley, S. C. (1998). "Relationship of trout recruitment to snowmelt runoff flows and adult trout abundance in six Colorado Mountain streams," *Rivers* 6, 240–250.

Lewin, J., and Brewer, P. A. (2001). "Predicting channel patterns," *Geomorphology* 40, 329–339.

Lewis Jr., W. M., Hamilton, S. K., Lasi, M. A., Rodriguez, M., and Saunders III, J. F. (2001). "A foodweb analysis of the Orinoco floodplain based on production estimates and stable isotope data," *Journal of the North American Benthological Society* 20, 241–254.

Lindegaard, C., Brodersen, K. P., Wiberg-Larsen, P., and Skriver, J. (1998). "Multivariate analysis of macrofaunal communities in Danish springs and springbrooks. *Studies in Crenobiology: The Biology of Springs and Springbrooks,* Botosaneanu, L., ed., Backhuys, Leiden, 201–219.

Lobon-Cervia, J. (2003). "Spatiotemporal dynamics of brown trout production in a Cantabrian stream: Effects of density and habitat quality," *Transactions of the American Fisheries Society* 132, 621–637.

López, F., and García, M. H. (1998). "Open-channel flow through simulated vegetation: Suspended sediment transport modelling," *Water Resources Research* 34(9), 2341-2352.

Loreau, M., Naeem, S., Inchausti, P., Bengtsson, J., Grime, J. P., Hector, A., Hooper, D. U., Huston, M. A., Raffaelli, D., Schmid, B., Tilman, D., and Wardle, D. A. (2001). Ecology—Biodiversity and ecosystem functioning: Current knowledge and future challenges. *Science* 294, 804–808.

Lott, D. (2001). "Ground beetles and rove beetles associated with temporary ponds in England," *Freshwater Forum* 17, 40–53.

Lott, D. A. (1996). "Beetles by rivers and the conservation of riparian and floodplain habitats," *Environmental Monitoring, Surveillance and Conservation using Invertebrates,* M. D. Eyre, ed., EMS Publications, Newcastle Upon Tyne, 36–41.

Lytle, D. A., and Poff, N. L. (2004). "Adaptation to natural flow regimes," *Trends in Ecology and Evolution* 19, 94–100.

MacArthur, R. H., and Wilson, E. O. (1967). *The Theory of Island Biogeography*, Princeton, New Jersey.

McCabe, D. J. (1998). "Biological communities in springbrooks," *Studies in Crenobiology: The biology of springs and springbrooks,* Botosaneanu, L., ed., Backhuys, Leiden, 221–228.

McGlynn, B. L., McDonnell, J. J., Seibert, J., and Stewart, M. K. (2003). "On the relationships between catchment scale and streamwater mean residence time," *Hydrological Processes* 17, 175–181.

Mäki-Petäys, A., Muotka, T., and Huusko, A. (1999). “Densities of juvenile brown trout (*Salmo trutta*) in two subarctic rivers: Assessing the predictive capability of habitat preference indices,” *Canadian Journal of Fisheries and Aquatic Sciences* 56, 1420–1427.

Malanson, G. P. (1993). “Riparian landscapes,” *Cambridge Study in Ecology*, Cambridge University Press.

Malcolm, I. A., Soulsby, C., Youngson, A. F., Hannah, D. M., Mclaren, I. S., and Thorne, A. (2004). “Hydrological influences on hyporheic water quality: Implications for salmon survival,” *Hydrological Processes* 18, 1521–1542.

Manderbach, R., and Reich, M. (1995). “Effects of dams and weirs on the ground beetle communities (Coleoptera, Carabidae) of braided sections of the Isar floodplain,” *Archiv für Hydrobiologie Supplement* 101, 573–588.

Manderbach, R., and Plachter, H. (1997). “Life strategy of the carabid beetle *Nebria picicornis* (Fabr. 1801) (Coleoptera, Carabidae) on river banks,” *Beitr. Ökol*. 3, 17–27.

Middleton, B. (1999). *Wetland Restoration, Flood Pulsing and Disturbance Dynamics*. John Wiley and Sons, Inc., New York.

Milner, A. M., Brittain, J. E. and Petts, G. E. (2001). “Trends of macroinvertebrate community structure in glacier-fed rivers in relation to environmental conditions: A synthesis,” *Freshwater Biology* 46, 1833–1848.

Mitro, M. G., Zale, A. V., and Rich, B. A. (2003). “The relation between age-O rainbow trout (*Oncorhynchus mykiss*) abundance and winter discharge in a regulated river,” *Canadian Journal of Fisheries and Aquatic Sciences* 60, 135–139.

Molofsky, J. (1994). “Population dynamics and pattern formation in theoretical populations,” *Ecology* 75, 30–39.

Moir, H. J. (1999). “The characterisation of salmon spawning habitat in rivers in NE Scotland,” Unpublished Ph.D diss., University of Aberdeen.

Moir, H., Soulsby, C., and Youngson, A., (1998). “Hydraulic and sedimentary characteristics of habitat utilized by Atlantic salmon for spawning in the Girnock burn, Scotland,” *Fisheries Management and Ecology* 5, 241–254.

Moir, H. J., Gibbins, C. N., Soulsby, C., and Webb, J. (2003). “Linking channel geomorphic characteristics to spatial patterns of spawning activity and discharge use by Atlantic Salmon,” *Geomorphology* 1441(60), 21–35.

Moir, H., Gibbins, C. N., Soulsby, C., and Youngson, A. (in prep.). “Validation of PHABSIM predictions for simulating salmon spawning habitat,” *Rivers Research and Application.*

Minshall, G. W. (1988). “Stream ecosystem theory: A global perspective,” *North American Benthological Society* 7(4), 263–288.

Moilanen, A., and Hanski, I. (1998). “Metapopulation dynamics: Effects of habitat quality and landscape structure.” *Ecology* 79, 2503–2515.

Montgomery, D. R. (1999). “Process domains and the river continuum,” *American Water Resources Association* 35, 397–410.

Mooney, H. A., and R. J. Hobbs (ed.) (2000). *Invasive Species in a Changing World,* Island Press, Washington DC.

Morris, T. J., and Corkum, L. D. (1996). "Assemblage structure of freshwater mussels (Bivalvia: Unionidae) in river with grassy and forested riparian zones." *Journal of the North American Benthological Society* 15;4, 576–586.

Murray, A. B., and Paola, C. (2003). "Modelling the effect of vegetation on channel pattern in bedload rivers," *Earth Surface Processes and Landforms* 28, 131–143.

Naiman, R. J., and Decamps, H. (1997). "The ecology of interfaces: Riparian zones," *Annual Review of Ecology and Systematics* 28, 621–658.

Naiman, R. J., Bunn, S. E., Nilsson, C., Petts, G. E., Pinay, G. and Thompson, L. C. (2002). "Legitimizing fluvial ecosystems as users of water," *Environmental Management* 30(4), 455–467.

Naiman, R. J., Decamps, H., Pastor, J., and Johnston, C. A. (1988). "The potential importance of boundaries to fluvial ecosystems," *Journal of the North America Benthological Society* 7, 289–306.

Naiman, R.J., Decamps, H., and Pollock, M. (1993). "The role of riparian corridors in maintaining regional biodiversity," *Ecological Applications* 3, 209–212.

Nakano, S., Miyasaka, H., and Kuhara, N. (1999). "Terrestrial-aquatic linkages: Riparian arthropod inputs alter trophic cascades in a stream food web," *Ecology* 80, 2435–2441.

Newbold, J. D., O'Neill, R. V., Elwood, J. W., and Van Winkle, W. (1982). "Nutrient spiralling in streams: Implications for nutrient limitations and invertebrate activity," *The American Naturalist* 120, 628–652.

Nienhuis, P. H., Buijse, A. D., Leuven, R. S. E. W., Smits, A. J. M., de Nooij, R. J. W., and Samborska, E. M. (2002). "Ecological rehabilitation of the lowland basin of the river Rhine (NW Europe)," *Ecological Restoration of Aquatic and Semi-aquatic Ecosystems in the Netherlands (NW Europe),* Nienhuis P. H., and R. D. Gulati ed., *Hydrobiologia* 478, 53–72.

Nienhuis, P. H., Chojnacki, J. C., Harms, O., Majewski, W., Parzonka, W., and Prus, T. (2000). Elbe, Odra and Vistula: reference rivers for the restoration of biodiversity and habitat quality. *New Approaches to River Management*, Smits A. J. M., Nienhuis P. H., and R. S. E. W. Leuven ed., Backhuys Publishers, Leiden.

Nilsson, C. (1996). "Remediating river margin vegetation along fragmented and regulated rivers in the north: What is possible?" *Regulated Rivers* 12, 415-431.

Nilsson, C., and Berggren, K. (2000). "Alterations of riparian ecosystems caused by river regulation," *BioScience* 50, 783–792.

Nilsson, C., and Svedmark, M. (2002). "Basic principles and ecological consequences of changing water regimes: Riparian plant communities," *Environmental Management* 30, 468–480.

Nilsson, C., Xiong, S., Johansson, M. E., and Vought, L.B.-M. (1999). "Effects of leaf-litter accumulation on riparianplant diversity across Europe." *Ecology* 80, 1770–1775.

O'Shea, D. T. (1995). "Estimating minimum instream flow requirements for Minnesota streams from hydrologic data and watershed characteristics," *North American Journal of Fisheries Management* 15, 569-578.

Olden, J., and Poff, N. L. (2003). "Redundancy and the choice of hydrologic indices for characterizing streamflow regimes," *River Research and Applic*ations 19, 101–121.

Olff, H., and Ritchie, M. E. (2002). "Fragmented nature: Consequences for biodiversity," *Landscape Urban Planning* 58, 83–92.

Opdam, P., Foppen, R., and Vos, C. (2002). "Bridging the gap between ecology and spatial planning in landscape ecology," *Landscape Ecology* 16, 767–779.

Osterkamp, W. R. (1998). "Processes of fluvial island formation, with examples from Plum Creek, Colorado and Snake River, Idaho," *Wetlands* 18, 530–545.

Pasternack, G. B., Wang, C. L. and Merz, J. E. (2004). "Application of a 2D hydrodynamic model to design of reseach-scale spawning gravel replenishment on the Mokelumne River, California," *River Research and Applications* 20, 205–225.

Petts, G. E. (1987). "Timescales for ecological change in regulated rivers," *Regulated Streams: Advances in Ecology*, Craig, J., and J. B. Kemper, ed., Plenum, New York, 257–266.

____________. (1996). "Water allocation to protect river ecosystems," *Regulated Rivers* 12, 353–367.

____________. (2000). "A perspective on the abiotic processes sustaining the ecological integrity of running waters," *Hydrobiologia* 422/423, 15–27.

____________. (2001). "Sustaining our rivers in crisis: Setting the international agenda for action," *Water Science Technology* 43, 3–16.

Petts, G. E., and Amoros, C. (ed.) (1996). *Fluvial hydrosystems.* Chapman and Hall, London.

Petts, G. E., and Gurnell, A. M. (in prep.). "Dams and geomorphology: Research progress and future directions," *Geomorphology*.

Petts G. E., and Maddock, I. (1994). "Flow allocation for in-river reeds," *The Rivers Handbook*, Calow, P., and Petts, G.E., ed., Blackwell Scientific, London, 289–307.

Petts, G. E., Bickerton, M. A., Crawford, C., Lerner, D. N., and Evans, D. (1999). "Flow management to sustain groundwater-dominated stream ecosystems," *Hydrological Processes* 13, 497–513.

Petts, G. E., Sparks, R., and Campbell, I. (2000). "River restoration in developed economies," *Global Perspectives on River Conservation: Science, Policy and Practice,* Boon, P. J., Davies, B. R., and Petts G. E., ed., John Wiley and Sons, 493–508.

Pinay, G., Clement, J. C., and Naiman, R. J. (2002). “Basic principles and ecological consequences of changing water regimes on nitrogen cycling in fluvial systems,” *Environmental Management* 30, 481–491.

Planty-Tabacchi, A. M., Tabacchi, E., Naiman, R. J., Deferrari, C., and Decamps, H. (1996). “Invasibility of species rich communities in riparian zones,” *Conservation Biology* 10, 598– 607.

Planty-Tabacchi, A. M., Tabacchi, E., and Salinas Bonillo, M. J. (2001). “Invasions of river corridors by exotic plant species: Patterns and causes,” *Plant invasions*, G. Brundu et al., ed., Klüwer publ., The Netherlands, 221–234.

Poff, N. L. (1992). “Why disturbance can be predictable: A perspective on the definition of disturbance in streams,” *Journal of North American Benthological Society* 11, 86–92.

____________. (1997). “Landscape filters and species traits: Towards mechanistic understanduing and prediction in stream ecology,” *Journal North American Benthological Society* 16(2), 391–409.

Poff, N. L., and Allan, D. (1995). “Functional organization of stream fish assemblages in relation to hydrological variability,” *Ecology* 76(2), 606–627.

Poff, N. L., and Ward, J. V. (1989). “Implication of streamflow variability and predictability for lotic community structure: A regional analysis of streamflow patterns,” *Canadian Journal of Fisheries Aquatic Science* 46, 1805–1818.

____________. (1990). “Physical habitat template of lotic systems: Recovery in the context of spatiotemporal heterogeneity,” *Environmental Management* 14, 629–645.

Poff, N. L., Allan, D. J., Bain, M. B., Karr, J. R., Prestegaard, K. L., Richter, B. D., Sparks, E. E., and Stromberg, J. C. (1997). “The natural flow regime—A paradigm for river conservation and restoration,” *Bio-Science* 47, 769–784.

Poff, N. L., Allan, J. D., Palmer, M. A., Hart, D. D., Richter, B. D., Arthington, A. H., Rogers, K. H., Meyer, J. L., and Stanford, J. A. (in prep). “River flows and water wars? Emerging science for environmental decision-making,” *Frontiers in Ecology and the Environment* 1(5).

Plachter, H. (1986). “Composition of the carabid beetle fauna of natural riverbanks and of man-made secondary habitats,” *Carabid Dynamics: Their Adaptations and Dynamics,* P. J. den Boer, M. L. Luff, D. Mossakowski, and F. Weber, ed., Gustav Fisher, Stuttgart, 509–535.

Plachter, H., and Reich, M. (1998). “The significance of disturbance for population and ecosystems in natural floodplains,” *The International Symposium on River Restoration, Tokyo*, 29–38.

Polis, G. A., Anderson, W. B., and Holt, R. D. (1997). “Toward an integration of landscape and food web ecology: The dynamics of spatially subsidized food webs,” *Annual Review of Ecology and Systematics* 28, 289–316.

Pollock, M. M., Naiman, R. J., and Hanley, T. A. (1998). "Predicting plant species richness in forested and emergent wetlands—a test of biodiversity theory," *Ecology* 79, 94–105.

Pouilly, M., and Souchon, Y. (1995). "Méthode des microhabitats: Validation et perspectives," *Bulletin Français de la Pêche et de la Pisciculture* 337/338/339, 329–336.

Preall, R. J., and Ringler, N. H. (1989). "Comparison of actual and potential growth rates of brown trout (*Salmo trutta*) in natural streams based on bioenergetic models," *Canadian Journal of Fisheries and Aquatic Sciences* 46, 1067–1076.

Pulliam, H. R. (1988). "Sources, sinks, and population regulation," *American Naturalist* 132, 652–661.

Pulliam, H. R., and Danielson, B. J. (1991). "Sources, sinks, and habitat selection—a landscape perspective on population-dynamics," *American Naturalist* 137, 50–66.

Railsback, S. F., and Harvey, B. C. (2003). "An individual-based trout model for instream flow assessment: Validity, advantages, and disadvantages," *International IFIM Users' Workshop*, Fort Collins, Colorado.

Rainer, M. W., Amon et al. (2003). "Dissolved organic carbon distribution and origin in the Nordic Seas: Exchanges with the Arctic Ocean and the North Atlantic," *Journal of Geophysical Research* 108(C7).

Rapport, D. J., Gaudet, C., Karr, J. R., Baron, J. S., Bohler Jackson, C., Jones, B., Naiman R. J., and Norton, B. (1998). "Evaluating landscape health: Integrating societal goals and biophysical goals," *Journal Environmental Management* 53, 1–15.

Reich, M. (1991). "Grasshoppers (Orthoptera, Saltatoria) on alpine and dealpine riverbanks and their use as indicators for natural floodplain dynamics," *Regulated Rivers* 6, 333–339.

Resh, V. H., Brown, A. V., Covich, A. P., Gurtz, M. E., Li, H. W., Minshall, G. W., Reice, S. R., Sheldon, A. L., Wallace, J. B., and Wissmar, R.C. (1988). "The role of disturbance in stream ecology," *Journal of North American Benthological Society* 7, 433–455.

Richards, K., Brasington, J., and Hughes, F. (2002). "Geomorphic dynamics of floodplains: Ecological implications and a potential modelling strategy," *Freshwater Biology* 47, 559–580.

Richter, B. D., Baumgartner, J. V., Powell, J., and Braun, P. D. (1996a). "A method for assessing hydrologic alteration within ecosystems," *Conservation Biology* 10, 1163–1174.

____________. (1996b). "A method for assessing hydrologic alteration within ecosystems," *Regulated Rivers* 14, 329–430.

Richter, B. D., Baumgartner, J. V., Wingington, R., and Braun, P. D. (1997). "How much water does a river need?" *Freshwater Biology* 37, 231–249.

Robinson, C. T., Tockner, K., and Ward, J. V. (2002). “The fauna of dynamic riverine landscapes,” *Freshwater Biology* 47, 661–677.

del Rosario, R. B., and Resh, V. H. (2000). “Invertebrates in intermittent and perennial streams: Is the hyporheic zone a refuge from drying,” *Journal of the North American Benthological Society* 19, 680–696.

Rose, K. A., and Cowan, Jr., J. H. (1993). “Individual-based model of young-of-the-year striped bass population dynamics. I. Model description and baseline simulations,” *Transactions of the American Fisheries Society* 122, 415–438.

Rosenberg, D. M., McCully, P., and Pringle, C. M. (2000). “Global-scale environmental effects of hydrological alterations: Introduction,” *BioScience* 50(9), 746–751.

Rouch, R., Mangin, A., Bakalowicz, M., and Hulst, D. (1997). “The hyporheic zone: Hydrogeological and geochemical study of a stream in the Pyrenees Mountains,” *Internaitonal Revue gesampte Hydrobiologie* 82(3), 357–378.

Sabaton, C., Siegler, L., Gouraud, V., Bagliniere, J. L., and Manne, S. (1997). “Presentation and first applications of a dynamic population model for brown trout (Salmo trutta L.): Aid to river management,” *Fisheries Management and Ecology* 4, 425–438.

Sabo, J. L., and Power, M. E. (2002). “River-watershed exchange: Effects of riverine subsidies on riparian lizards and their terrestrial prey,” *Ecology* 83, 1860–1869.

Sadler, J. P., Bell, D., and Fowles, A. P. (2004). “The hydroecological controls and conservation value of beetles on exposed riverine sediments in England and Wales,” *Biological Conservation* 118, 41–56.

Salo, J., Kalliola, R., Häkkinen, I., Mäkinen, Y., Niemelä, P., Puhakka, M., and Coley, P. D. (1986). “River dynamics and the diversity of the Amazon lowland forest,” *Nature* 322, 254–258.

Sanchez, E., Bravo de la Parra, R., and Auger, P. (1995). “Linear discrete models with different time scales,” *Acta Biotheoretica* 43, 465–479.

Sauvage, S., Teissier, S., Vervier, P., Ameziane, T., Garabetian, F., Delmas, F., and Caussade, B. (2003). *River Research and Applications* 19, 181–198.

Scheibe, T. D., and Richmond, M. C. (2002). “Fish individual-based numerical simulator (FINS): A particle-based model of juvenile salmonid movement and dissolved gas exposure history in the Columbia river basin,” *Ecological Modelling* 147, 233–252.

Scruton, D. A., and Ledrew, L. J. (1997). “A retrospective assessment of the flow regulation of the West Salmon river, Newfoundland, Canada,” *Fisheries Management and Ecology* 4, 467–480.

Sedell, J. R., and Froggatt, J. L. (1984). “Importance of streamside forests to large rivers: The isolation of the Willamette River, Oregon, U.S.A., from its floodplain by snagging and streamside forest removal: Verhandlungen,” *Internationale Vereinigung Fur Theoretische Und Angewandte Limnologie* 22(3), 1828–1834.

Sih, A., Jonsson. B. G., and Luikart, G. (2000). "Habitat loss: Ecological, evolutionary and genetic consequences," *Trends in Ecology and Evolution* 15, 132–134.

Simberloff, D. (1998). "Flagships, umbrellas, and keystones: Is single-species management passé in the landscape era?" *Biological Conservation* 83, 247–257.

Sinsabaugh, R. L., and Findlay, S. E. G. (2003). "Dissolved organic matter: Out of the black box and into the mainstream," *Aquatic Ecosystems: Interactivity of DOM*, 479–488.

Shirvell, C. S. (1994). "Effect of changes in streamflow on the microhabitat use and movements of sympatric juvenile coho salmon (*Oncorhynchus kisutch*) and chinook salmon (*O. tshawytscha*) in a natural stream," *Canadian Journal of Fisheries and Aquatic Sciences* 51, 1644–1652.

Sket, B. (1999). "The nature of biodiversity in hypogean waters and how it is endangered," *Biodiversity and Conservation* 8, 1319–1338.

Smits, A. J. M., Havinga, H., and Marteijn, E. C. L. (2000). "New concepts in river and water management in the Rhine River basin: How to live with the unexpected?" *New Approaches to River Management,* Smits A. J. M, Nienhuis P. H. , Leuven R. S. E. W., ed., Leiden, Backhuys Publishers, 267–286.

Smith, H. (2000). "The hydro-ecology of limestone springs in the Wye Valley, Derbyshire," unpublished Ph.D diss., University of Huddersfield, UK.

Smith, H., and Wood, P. J. (2002). "Flow permanence and macroinvertebrate community variability in limestone spring systems," *Hydrobiologia* 487, 45–58.

Smits, A. J. M., Nienhuis, P. H., and Leuven, R. S. E. W. (ed.) (2000). *New Approaches to River Management*, Backhuys, Leiden.

Soulsby, C., Chen, M., Ferrier, R. C., Jenkins, A., and Harriman, R. (1998). "Hydrogeochemistry of shallow groundwater in a Scottish catchment," *Hydrological Processes* 12, 1111–1127.

Soulsby, C., Malcolm, I. A., Hannah, D. M., and Youngson, A. Y. (in prep.). "Flow variability in an upland stream: Ecological implications."

Southwood, T. R. E. (1977). "Habitat, the template for ecological strategies?" *Journal of Animal Ecology* 46, 337–365.

Stalnaker, C. B., and Arnette, S. C. (1976). "Methodologies for the determination of stream resource flow requirements: An assessment," U.S. Fish and Wildlife Services, Office of Biological Services Western Water Association.

Stanford, J. A., and Ward, J. V. (1993). "An ecosystem perspective of alluvial rivers: Connectivity and the hyporheic corridor," *Journal of the North American Benthological Society* 12, 48–60.

Stanford, J. A., Ward, J. V., Liss, W. J., Frissell, C. A., Williams, R. N., Lichatowich, J. A., and Coutant, C. C. (1996). "A general protocol for restoration of regulated rivers," *Regulated Rivers: Research and Management* 12, 391–413.

Statzner, B., and Higler, B. (1986). "Stream hydraulics as a major determinant of benthic invertebrate zonation patterns," *Freshwater Biology* 16, 127–139.

Statzner, B., and Resh, V. H. (1993). "Multiple site and -year analysis of stream insect emergence : A test of ecological theory," *Oecologi* 96, 65–79.

Stelter, C., Reich, M., Grimm, V., and Wissel, C. (1997). "Modelling persistence in dynamic landscapes: Lessons from a metapopulation of the grasshopper *Bryodema tuberculata*," *Journal of Animal Ecology* 66, 508–518.

Stewardson, M., and Gippel, C. (1997). "In-stream environmental flow design: A review," draft report, Cooperative Research Centre for Catchment Hydrology, Department of Civil and Environmental Engineering, University of Melbourne, Vicktoria, Australia.

Stoesser, T., Wilson, C. A. M. E., Bates, P. D., and Dittrich, A. (2003). "Application of a 3D numerical model to a river with vegetated floodplains," *Journal of Hydroinformatics* 5, 99–112.

Studley, T. K., Baldrige, J. E., and Railsback, S. F. (1996). "Predicting fish population response to instream flows," *Hydro Review* 15, 48–57.

Tabacchi, E., and Planty-Tabacchi, A. M. (2003). "Recent changes in riparian vegetation: Possible consequences on dead wood processing along rivers," *River Research and Applications* 19, 251–263.

____________. (2001). "Functional significance of species composition in riparian plant communities," *Journal American Water Resource Association* 37(6), 1629–1637.

Tabacchi, E., Correll, D. L., Hauer, R., Pinay, G., Planty-Tabacchi, A. M., and Wissmar, R. C. (1998). "Development, maintenance and role of riparian vegetation in the river landscape," *Freshwater Biology* 40, 497–516.

Tennant, D. L. (1976). "Instream flow regimens for fish, wildlife, recreation and related environmental resources," *Fisheries* 1(4), 6–10.

Tharme, R. E. (2003). "A global perspective on environmental flow assessment: Emerging trends in the development and application of environmental flow methodologies for rivers," *River Research and Applications* 19, 397–442.

Thomas, C. D., and Kunin, W. E. (1999). "The spatial structure of populations," *Journal of Animal Ecology* 68, 647–657.

Thorp, J. H., and Delong, M. D. (2002). "Dominance of autochthonous autotrophic carbon in food webs of heterotrophic rivers," *Oikos* 96, 543–550.

____________. (1994). "The Riverine Productivity Model—an heuristic view of carbon—sources and organic-processing in large river ecosystems," *Oikos* 70, 305–308.

Thorp, J. H., Delong, M. D., Greenwood, K. S., and Casper, A. F. (1998). "Isotopic analysis of three food web theories in constricted and floodplain regions of a large river," *Oecologia* 117, 551–563.

Tischendorf, L. (2001). “Can landscape indices predict ecological processes consistently?” *Landscape Ecology* 16, 235–254.

Tockner, K., and Stanford, J. A. (2002). “Riverine flood plains: Present state and future trends,” *Environmental Conservation* 29, 308–330.

Tockner, K., Bunn, S. E., Quinn, G., Naimann, R., Stanford, J. A., and Gordon, C. (in prep. a). “Flood plains: Critically threatened ecosystems,” in N. C. Polunin, ed., *The State of the World’s Ecosystems*, Cambridge University Press.

Tockner, K., Claret, C., and Zettel, J. (in prep. b). “Ecology of braided rivers,” *Proceedings of the Braided River Symposium, Birmingham, April 2003*.

Tockner, K., Malard, F., and Ward, J. V. (2000). “An extension of the flood pulse concept,” *Hydrological Processes* 14, 2861–2883.

Tockner, K., Paetzold, A., Karaus, U., Claret, C., Zettel, J. (in prep. c). “Ecolory of braided rivers,” in G. H. Sambrook-Smith, J. L. Best, C. S. Bristow, and G. Petts, ed., *Braided Rivers – IAS Special Publication*, Blackwell, Oxford.

Tockner, K., Ward, J. V., Arscott, B. A., Edwards, P. J., Kollmann, J., Gurnell, A. M., Petts, G. E., and Maiolini, B. (2003). “The Tagliamento River: A model ecosystem of European importance,” *Aquatic Sciences* 65, 239–253.

Tockner, K., Ward, J. V., Edwards, P. J., and Kollmann, J. (2002). “Riverine landscapes: An introduction,” *Freshwater Biology* 47, 497–500.

Townsend, C. R. (1989). “The patch dynamic concept of stream community ecology,” *Journal North American Benthological Society* 8(1), 36–50.

Townsend, C. R., and Riley, R. H. (1999). “Assesment of river health: Accounting for perturbation pathways in physical and ecological space,” *Freshwater Biology* 41, 393–404.

van der Nat, D., Tockner, K., Edwards, P. J., Ward, J. V., and Gurnell, A. M. (2003). “Habitat change in braided flood plains (Tagliamento, NE-Italy),” *Freshwater Biology* 48, 1799–1812.

Van Winkle, W., Jager, H. I., Railsback, S. F., Holcomb, B. D., Studley, T. K., and Baldrige, J. E. (1998). “Individual-based model of sympatric populations of brown and rainbow trout for instream flow assessment: Model description, sensitivity analysis, and calibration,” *Ecological Modelling* 110, 175–207.

Van Winkle, W., Rose, K. A., and Chambers, R. C. (1993). “Individual-based approach to fish population dynamics: An overview,” *Transactions of the American Fisheries Society* 122, 397–403.

Vanotte, R. L., Minshall, G. W., Cummins, K. W., Sedell, J. R., and Cushing, C. E. (1980). “The river continuum concept,” *Canadian Journal of Fisheries and Aquatic Sciences* 37, 130–137.

Von Manderbach, R., and Reich, M. (1995). “Effects of dams and weirs on the ground beetle communities (Coleoptera, Carabidae) of braided sections of the Isar floodplain,” *Archiv für Hydrobiologie Supplement* 101, 573–588.

Walsh, S. J., Butler, D. R., and Malanson, G. P. (1998). “An overview of scale, pattern, process relationships in geomorphology: A remote sensing and GIS perspective,” *Geomorphology* 21, 183–205.

Ward, J. V. (1989). “The four-dimensional nature of lotic ecosystems,” *Journal of the North American Benthological Society* 8, 2–8.

____________. (1998). “Riverine landscapes: Biodiversity patterns, disturbance regimes, and aquatic conservation,” *Biological Conservation* 83, 269–278.

Ward, J. V., and Stanford, J. A. (1983). “The intermediate-disturbance hypothesis: An explanation for biotic diversity patterns in lotic ecosystems,” *Dynamics of lotic ecosystems,* Fontaine, T. D., and Bartell, S. M. ed., Ann Arbor, Michigan, Ann Arbor Science, 347–356.

____________. (1995). “Ecological connectivity in alluvial river ecosystems and its disruption by flow regulation,” *Regulated Rivers* 11, 105–119.

Ward, J. V., and Tockner, K. (2001). “Biodiversity: Towards a unifying theme for river ecology,” *Freshwater Biology* 46, 807–819.

Ward, J. V., Malard, F., and Tockner, K. (2001). “Landscape ecology: A framework for integrating pattern and process in river corridors,” *Landscape Ecology* 17, 35–45.

Ward, J. V., Tockner, K., Arscott, D. B., and Claret, C. (2002). “Riverine landscape diversity,” *Freshwater Biology* 47, 517–539.

Ward, J. V., Tockner, K., Edwards, P. J., Kollmann, J., Bretschko, G., Gurnell, A. M., Petts, G. E., and Rossaro, B. (1999). “A reference river system for the Alps: The Fiume Tagliamento,” *Regulated Rivers* 15, 63–75.

Watkinson, A. R., and Sutherland, W. J. (1995). “Sources, sinks and pseudo-sinks,” *Journal of Animal Ecology* 64, 126–130.

WCW. (2000). “Dams and development.” World Commission on Dams, Earthscan, London.

Webb, J. H., Gibbins, C. G., Moir, H., and Soulsby, C. (2001). “Flow requirements of spawning Atlantic salmon in an upland stream: Implications for water resource management,” *Journal of the Chartered Institute of Water and Environmental Management* 15, 1–8.

Welsch, D. J. (1991). “Riparian forest buffers: Functions and design for protection and enhancement of water resources,” USDA Forest Service, Northeastern Area State and Private Forestry, Forest Resources Management, Radnor, Pennsylvania.

Welsch, D. J., Hornbeck, J., Verry, E. S., Dolloff, C. A., and Greis, J. G. (2000). “Riparian area management: Themes and recommendations,” *Riparian management in the eastern continental United States,* E. S. Verry, J. Hornbeck, and C. A. Dolloff, ed., CRC-Lewis Publishers, Florida.

Whittaker, D., and Shelby, B. (2000). “Managed flow regimes and resources values: Traditional versus alternative strategies,” *Rivers* 7, 233–244.

Wiens, J. A. (1997). “Metapopulation dynamics and landscape ecology.” *Metapopulation biology: Ecology, genetics and evolution*, M. Gilpin and I. Hanski, ed., Academic Press, 43–62.

Williams, D. D. (1987). *The ecology of temporary waters.* Croom Helm, Routledge, Chapman and Hall, London.

Williams, D. D., and Williams, N. E. (1998). “Invertebrate communities from freshwater springs: What can they contribute to pure and applied ecology,” *Studies in crenobiology: The biology of springs and springbrooks,* L. Botosaneanu, ed., Backhuys, Leiden, 251–261.

Williams, J. G. (1999). “Stock dynamics and adaptive management of habitat: An evaluation based on simulations,” *North American Journal of Fisheries Management* 19, 329–341.

Williamson, S. C., Bartholow, J. M., and Stalnaker, C. B. (1993). “Conceptual model for quantifying pre-smolt production from flow-dependent physical habitat and water temperature,” *Regulated Rivers* 8, 15–28.

Wolfert, H. P. (2001). “Geomorphological change and river rehabilitation: Case studies on lowland fluvial systems in the Netherlands,” *Alterra Scientific Contributions 6*, Wageningen.

Wood, P. J., Agnew, M. D., and G. E. Petts (2000). “Flow variations and macroinvertebrate community responses in a small groundwater-dominated stream in south-east England,” *Hydrological Processes* 14, 3133–3147.

Wood, P. J., Gunn, J., Smith, H., and Abris-Kutty, A. (in prep.). “Flow permanence and macroinvertebrate community diversity within groundwater dominated headwater streams and springs, “ *Hydrobiologic.*

Youngson, A. F., Buck, R., Simpson, T. H., and Hay, D. W. (1983). “The autumn and spring emigrations of juvenile Atlantic salmon, Salmo-salar from the Girnock burn, Aberdeenshire, Scotland—Environmental release of migration,” *Journal of Fish Biology* 23(6), 625–639.

Zalewski, M. (ed.) (2002). “Guidelines for the integrated management of the watershed—polytechnology and ecohydrology,” *UNEP– ETC Freshwater Management Series* 5, Osaka, Japan.

Zerm, M., and Adis, J. (2003). “Exceptional anoxia resistance in larval tiger beetle, Phaeoxantha klugii (Coleoptera : Cicindelidae),” *Physiological Entomology* 28, 150–153.

Zulka, K. P. (1994). “Carabids in a Central European floodplain: Species distribution and survival during inundations.” *Carabid Beetles: Ecology and evolution.* K. Desender, M. Dufêne, M. Loreau, M. L. Luff, and J.-P. Maelfait, ed., Kluwer, Dordrecht, 399–405.

Appendix A
Program and Participants

Program

Tuesday 16th September

12.30 Check-in available at Lucas House, Edgbaston Park Road, University of Birmingham

14.15 Introduction, welcome, aims, targets and outputs. *Geoff Petts*

14.30 Do we really need more science? Should we make better use of what we know already? Experience from the UK. *Hugh Sambrook and Geoff Petts*.

15.15 – 16.30 **Management and Planning Perspectives: England and Wales**
Water resources assessment and abstraction management in England and Wales. *John Aldrick.*
The Environment Agency's approach to integrating ecology and hydrology. *Juliette Hall*.
Climate change and uncertainty in water resources planning and river management. *Merylyn Hedger.*

16.45 – 18.30 Short orientation tour of the Edgbaston Campus: Winterbourne Gardens, Barber Institute (Art Gallery), Great Hall, Lapworth Museum (Geology), University Station (Rail), Staff House (Bar, cafeteria and shops) followed by dinner at 1845.

19.45 – 20.30 **Landscape- scale science**
The role of landscape ecology in river management. *Piet Nienhuis.*

Wednesday 17th September.: Critical Reviews of Key Issues

09.00 – 10.30 **Environmental flows**
Environmental flows and hydropower. *John Brittain*.

Flow variability and stream ecosystem evolution. *Pascal Breil.*

Flow regimes and ecological integrity of headwater streams. *Chris Soulsby*.

10.50 – 12.20 **Core Habitats of 21st Century Rivers**?
Ecological functioning of temporary headwater streams and springs. *Paul Wood*.

Ecological functioning of exposed riverine sediment 'islands'. *Jon Sadler*.

Ecological dynamics of narrow, linear, riparian systems. *Eric Tabacchi.*

13.45 – 15.15 **Water Quality and Riverine Productivity in Highly-populated Catchments**
Water quality and water resources management. *John Martin*.

Riverine productivity models. *Mark Ledger*.

Dissolved organic matter in aquatic systems. *Andy Baker*.

15.30 – 17.30 **Linking Physical and Ecological Models**
Patterns and processes in river corridors. *Klement Tockner*.

Floodplain modelling of sedimentation and vegetation development. *Martin Baptist*.

Modelling fish population dynamics and instream flows. *Herve Capra*.

19.00 – Dinner followed by
20.00 – 21.30 Informal Meetings of Thursday's Discussion Groups.

The Elements of a New Integrated Model.
Discussion led by *John Nestler* and *Herve Capra,* Rapporteur: *Chris Soulsby.*
Other group members: *Martin Baptist, Meryln Hedger, Geoff Petts, Hugh Sambrook,*

Ecological Principles for Sustainable Water Resources Management.
Discussion led by *Barb Kleiss* and*Piet Nienhuis.* Rapporteur *John Brittain,*
Other group members: *Malcolm Greenwood, Juliette Hall, Mark Ledger, Eric Tabacchi, Pascal Breil*

A Roadmap for Research and Data Needs.
Discussion led by *Jean O'Neil,* and *Klement Tockner.* Rapporteur *Andy Baker*.
Other group members: *John Aldrick, Melanie Bickerton, Robert Kennedy, John Martin.*

Thursday 18th September: A Forward Look.
Chair: Geoff Petts

09.00 – 10.15 **The Elements of a New Integrated Model**. Discussion led by *John Nestler* and *Herve Capra,* Rapporteur: *Chris Soulsby.*

10.45 – 12.00 **Ecological Principles for Sustainable Water Resources Management.** Discussion led by *Barb Kleiss* and*Piet Nienhuis.* Rapporteur *John Brittain,*

Chair Bob Kennedy
13.00 – 14.15 **A Roadmap for Research and Data Needs.** *Discussion led by Jean O'Neil,* and *Klement Tockner.* Rapporteur Andy Baker.

14.15 – 15.00 **Summary, Conclusions.** Geoff Petts.

Participants

John Aldrick

Dr. John Aldrick, Water Resources Regulation Manager, The Environment Agency, Rivers House, 21 Park Square South, Leeds LS1 2QG
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John's responsibilities cover the Abstraction Licensing Process by which we manage Water Resources, and also Catchment Abstraction Management Strategies (CAMS) and Resource Assessment and Management (RAM), the technical methodology for water resource assessment. I have been involved with the development of CAMS and RAM, which have been launched as an Agency-wide framework for Water Resource Management.

Andy Baker

Dr. Andy Baker
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Research interests include water quality, river pollution, and environmental biogeochemical techniques. He has developed the use of dissolved organic matter fluorescence as a tracer of both natural and anthropgenic organic matter in rivers. He has won numerous research grants and has published over 50 articles in internationally refereed journals including *Nature*, *Environmental Science and Technology*, *Journal of Hydrology*, *Hydrological Processes and Water Research*.

Martin Baptist

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Martin studied integrated water management, with an emphasis on modelling aquatic ecology, at the Wageningen Agricultural University. He graduated in 1996 and started working at Delft Hydraulics. He has a broad understanding of the functioning of (eco)systems, governing the hydro- and morphodynamics, water quality and ecology. Since 2000, Martin has specialized in biogeomorphology of river floodplains, using mathematical models to predict the consequences of nature restoration on the flow of water, morphodynamics, and vegetation development.

Pascal Breil

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Pascal is a hydrologist with particular interests in flow variability and time-series analyses and the analyses of 'event sequences' in relation to the development of stream ecosystems and the management of flow variability with particular reference to fish populations.

John Brittain

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At present seconded to head an R & D programme on environmental flows at the Norwegian Water Resources and Energy Directorate. Adjunct Professor at the Department of Nature Conservation, Agricultural University of Norway. Main interests are the environmental effects of hydropower, aquatic entomology, and arctic and alpine freshwaters.

Hervé Capra

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Hervé is a fish ecologist with a biological/ecological university training (with special emphasis on river ecosystems) at University of Lyon. His main topics are stream physical habitat description, fish habitat relationships, and, more recently, fish population dynamics modelling (including habitat limiting events).

Juliette Hall

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Juliette has an academic background in Environmental Science and Applied Ecology. The main focus of her current post is to provide the link internally between the Water Resources function and the Conservation and Ecology and Fisheries functions to improve our understanding of hydro-ecological relationships and externally with conservation agencies (e.g., English Nature and Countryside Council for Wales) and DEFRA (Department for Environment, Food and Rural Affairs) to produce policies and guidance that ensure that our environmental obligations are fully integrated in day-to-day water resources activities.

Merylyn Hedger

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Merylyn joined the Environment Agency in 2002, having been Director of the UK Climate Impacts Programme and, prior to that, International Co-ordinator of WWF's Climate Change Campaign. She started her professional life as town planner in Wales and has worked overseas in Latin America, South Asia, and the Pacific. Merylyn studied at the London School of Economics and Liverpool University and holds a Ph.D. in energy policy from Imperial College, London. In 2002 she was awarded the OBE for services to climate change assessment. She is a Board member of the Tyndall Centre.

Robert Kennedy

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Robert received a Ph.D. degree in 1978 from Kent State University, Kent, Ohio. He is a Research Limnologist with 25 years research experience in eutrophication, reservoir limnology, lake and reservoir restoration, and water quality and watershed management. The ERO’s mission is to link overseas researchers and research programs with similar ERDC research efforts.

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BS, MS, and Ph.D. from Spring Hill College, University of Southern Mississippi, and Louisiana State University. Research responsibilities involve wetlands functional assessment, the development of hydrogeomorphic concepts for ecological assessment work, and the application of ecological assessment tools across spatial and temporal scales to develop functional planning and management tools for watershed restoration.

Mark Ledger

Dr. Mark E. Ledger, School of Geography, Earth and Environmental Sciences, The University of Birmingham, Edgbaston, Birmingham B15 2TT
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I am a freshwater ecologist based in the School of Geography, Earth and Environmental Sciences, University of Birmingham. My research encompasses invertebrate and algal biodiversity, trophic interactions, and disturbance ecology. I am particularly interested in the effects that acidification and low flows have on the structure and function of stream communities. My current work relates increased drought frequency to secondary production and energy flow through aquatic food webs in experimental streams

John Martin

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John graduated from London University in 1971 with a B.Sc. in Chemistry and gained his Masters in Applied Hydrobiology in 1974. He lectured at Chelsea College London (now incorporated into Kings College) on aquatic chemistry and water quality modelling from 1974 to 1978 when he moved to Severn Trent Water Authority as a quality modeller and statistician. Since then, he has worked for Severn Trent on water quality and waste water regulation including the implementation of EU Directives and asset management planning. John contributes to national water quality policy debates via Water UK and is also an active member of the Chartered Institution of Water and Environmental Management (CIWEM).

John Nestler

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M.S., Zoology, University of Georgia,1976; and Ph.D., Zoology, Clemson University, 1980. Research interests: More realistically portray ecosystem-level processes and address sustainable development issues by integrating biological models with hydraulic/hydrologic models.

Piet Nienhuis

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Piet has 38 years of international research experience in marine biology, aquatic botany, estuarine and river management, and environmental science, employed by the Dutch Royal Academy of Sciences and, from 1988, by the University of Nijmegen, the Netherlands. His present focus is the ecological basis for river management. He published over 200 papers and books. He held several functions in international science management.

Jean O'Neil

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BS, MS, and Ph.D from Iowa State University, Southern Illinois University, and Texas A&M. Research interests are to translate ecosystem work across spatial and temporal scales and to develop functional planning and management tools for end users in system-wide restoration. Models of interest are index and conceptual.

Geoff Petts

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His research addresses the interface of the physical and biological sciences. His major focus has been on regulated rivers, the ecological impacts of flow regulation and water abstraction, and the determination of environmental flows. The research advances three themes: physical characteristics of rivers, especially thermal regimes and energy budgets and siltation processes; interactions between physical processes, ecological processes and biological communities; and applications to river management. He is also Editor in Chief of the journal *River Research and Applications*, formerly Regulated Rivers, and has authored 35 official reports for the Environment Agency, Conservation Agencies and the Water Companies.

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His research interests are the ecology and palaeoecology of insect species. His current research is supported by the Research Councils and The Environment Agency, Countryside Council for Wales and English Nature and examines the functioning of riparian (especially Exposed Riverine Sediments) and urban Biodiversity in the UK and elsewhere.

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B.A. Northumbria. Ph.D. Swansea. Chris is a hydrologist with interests in the hydroecology of upland, salmonid streams, hydraulic habitats, and surface water and groundwater interactions. Current research extends to the influence of hydrological disturbance events on stream ecology. Currently, he is a member of the Advisory Board of the (UK) River Restoration Centre.

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Eric is a stream ecologist currently involved in the relationships between riparian plant diversity, spatial heterogeneity and disturbance regimes along temperate rivers. As CNRS scientist researcher, he is involved in or leading several projects concerning plant invasions along streams, consequences on global warming on riparian plant diversity, and the effect of landscape fragmentation and flood regulation on riparian plant succession. During these last years, Eric has spent much time on plant dispersal and seed bank processes along river corridors and on the relationships existing between riparian community dynamics and the stability of riverbeds, including hydraulic and geomorphological aspects. Eric is co-leader of the team "Diversity and Functions of Riparian Communities" of the LADYBIO laboratory.

Klement Tockner

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Klement is a freshwater ecologist with special emphasis on river-wetland ecosystems and biodiversity. In particular, he is interested in the relationship between habitat heterogeneity, functional processes, and species diversity patterns in riverine flood plains. He is coordinating a major interdisciplinary research program on the Fiume Tagliamento (Italy), the last intact river corridor in the Alps. Recently he started a project to develop indicators for assessing the ecological integrity of river corridors. A main aspect is to link aquatic and terrestrial ecology. He is also a subject Editor of *Aquatic Sciences* and *Arch. Hydrobiol. Suppl. Large Rivers*.

Paul Wood

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Paul is a hydro-ecologist whose research has focussed on the influence of environmental variability on the aquatic invertebrate communities of groundwater dominated systems (streams, headwater springs, and subterranean cave systems). His research is centered on three areas: the role and influence of flow / discharge variability on macroinvertebrate communities, the interactions between physical processes and aquatic ecology, and the management of human impacted systems.

Other Attendees (Hydro-ecology Research Group, University of Birmingham)

Melanie Bickerton (Freshwater ecology)	M.A.Bickerton@bham.ac.uk
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14. ABSTRACT

The key to successful water and river management is the advancement of holistic approaches that seek to benefit human societies by sustaining the full range of resources created by rivers, including both physical and ecological services. This report describes the results of discussions held at the University of Birmingham, UK, during which participants sought to fill the conceptual gap that exists among water resource planners, flood engineers, and ecologists. Participants, including experts from Europe and the U.S. Army Corps of Engineers, attempted to advance and integrate concepts related to reference systems and sustainability and related to fully integrated water resource management within and between river basins. In a context of increasing pressures on (a) water supplies, wastewater treatment, and needs for flood management, (b) agricultural and forestry production systems, (c) land for urban expansion, and (d) nature conservation, recreation, and landscape restoration, participants discussed the primary challenge of managing changing rivers (changing flows, mobile sediments, and moving channels) in a diverse, dynamic, and highly connected system.

15. SUBJECT TERMS

Landscape ecology	Models	Rivers
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