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Are We Meeting the Challenges
of Landscape-Scale Riverine Research?
A Review

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Abstract

Identifying and quantifying relationships among landscape patterns, anthropogenic disturbances, and aquatic ecosystems is a new and rapidly developing approach to riverine ecology. In this review, we begin by describing the policy and management drivers for landscape-scale riverine research and we synthesize the technological advances that have enabled dramatic progress in the field. We then describe the development of landscape-scale riverine research through a series of landmark theoretical and review papers. Focusing on landscape-fish relationships, we consider the degree to which past efforts have been successful at meeting three challenges: (1) Has new research effectively incorporated the strengths of new technologies or are we doing the same old thing with more expensive data? (2) Have we incorporated key concepts from landscape ecology to improve our understanding of how landscapes affect rivers? (3) Have we been able to use landscape analyses to address management and policy needs? We conclude with a review of opportunities for advancement in the field of landscape-scale riverine research. These include moving toward the development of mechanistic theories of how landscapes affect rivers across disparate regions; considering the spatio-temporal structure of human impacts to landscapes; harnessing new statistical tools; and carefully defining landscape and response metrics to capture specific features.

Keywords: Aquatic ecosystems, Riverine ecology, Landscape patterns, River management, Catchment scale, Watershed scale

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1   Introduction

Riverine networks are intricately linked to the landscapes through which they flow (Hynes, 1975) (see Appendix A) and it is widely accepted that riverine management should occur over whole watersheds (Naiman *et al.*, 1992; Frissell *et al.*, 1986). Many fundamental theories and models in river ecology have their bases in landscape-scale thinking. While some approaches are naturally borrowed from the field of landscape ecology (see Section 4.2), we assert that much of the research linking landscapes to rivers has arisen relatively independently. For example, the river continuum concept (Vannote *et al.*, 1980) considers changes in carbon processing and biota with increasing drainage area across entire riverscapes. Sediment routing and nutrient models (see Zweimüller *et al.*, 2008; Tetzlaff *et al.*, 2007) incorporate vegetation type, soil conditions, land use, and lithology over large areas. In this review, we focus on how landscape influences on rivers, and in particular on fish assemblages, have been and can be measured, quantified, and applied to freshwater management problems; we draw on examples from multiple continents.

In Section 2 of this review, we summarize the policy and management drivers for landscape-scale riverine research, remind readers of the ecological drivers, and synthesize the technological advances that have enabled dramatic progress in the field. In Section 3, we describe the development of landscape-scale riverine research through a series of landmark theoretical and review papers. In our Section 4, we consider three important questions with respect to landscape-scale riverine research:

- Has new research effectively incorporated the strengths of new technologies or are we doing the same old thing with more expensive data?

- Have we incorporated key concepts from landscape ecology to improve our understanding of how landscapes affect rivers?

- Have we been able to use landscape analyses to address management and policy needs?

We synthesize existing literature to answer these questions and, in so doing, review progress to date. In Section 5, we use our review to identify opportunities for future efforts.

The roots of landscape-scale riverine research lie not in process-based models or mechanistic research, as in watershed ecology, but in the evaluation of pattern using statistical or correlative models, guided by mechanistic principles. Such analyses necessarily draw heavily on concepts from both river ecology and landscape ecology. Ideas commonly explored in these analyses include correlations between particular land uses and instream conditions, the impact of geographic extent on the strength of these correlations, and the degree to which local relationships between predictor and response variables at a large numbers of sites can be aggregated at larger scales. We define scale as the grain (resolution) and extent of both the predictor variables (generally measures of landscape condition) and the response variables (measures of instream condition and biological response).

We differentiate landscape-scale riverine research from traditional river ecology, riparian ecology, or watershed ecology by (i) the large spatial extent of the analyses, (ii) the use of spatial databases, and (iii) the emphasis on how landforms and land use outside of the active channel or the riparian area affect streams and rivers (Figure 1). We limit ourselves in this review to research that includes all three of the above features and to projects that consider fish as the biological response of interest. Except for a few examples, we exclude the diverse and important research concerning landscape-scale impacts on other instream biological responses, on physical and chemical habitat, and on terrestrial or marine biota.

Decreasing Spatial Extent →

System Attribute (anthropogenic action)	River basin (>10,000 km²)	Watershed (100–10,000 km²)	Stream (10–100 km²)	Reach (<10 km²)	Habitat (<1 km²)
Physical					
Altered land cover	Modified riparian species	Reduced riparian cover	Reduced wood loading	Reduced habitat complexity	Fewer pools; increased fines
Hydromodification[1]	Altered spatial connections		Less available/suitable habitat for mobile species		Lower habitat heterogeneity
Hydrological					
Altered land cover	Increased overland runoff	Increased peak flow	Altered flow and thermal regimes; reduced sediment supply		Increased scour
Hydromodification[1]	Increased inter-basin mixing	Altered hydrologic regime	Reduced lateral connections; isolation of off-channel habitats		Reduced refuge habitat
Water quality					
Altered land cover	Increased surface erosion and runoff (sediment, nutrients, toxics)			Degraded water chemistry	Locally intolerable conditions
Hydromodification[1]	Localized modifications in water chemistry/temperature		Reduced dissolved oxygen; increased temperatures		Locally intolerable conditions
Biological					
Altered land cover	Reduced species diversity		Increased abundances of tolerant species		
Hydromodification[1]	Increased non-native species and disease vectors		Altered encounter rates (affects biological interactions)		Shifts in aquatic comunity

Figure 1: Anthropogenic impacts, including altered land cover and hydromodification, drive physical, hydrological, chemical, and biological riverine response at the basin scale; these changes trickle down through the watershed scale, the stream scale and finally to reach-scale impacts on habitat and biota.
[1] Hydromodification includes anything that modifies/disrupts the flow of water or aquatic animals (i.e., dams, culverts, weirs, armored banks, levees, ditches).

2   Drivers of large-scale riverine research

2.1   Policy drivers

Management and conservation of aquatic habitats and their biota are often subject to new or adapted legal frameworks that rely on large-scale analyses. These legal frameworks may result from multiple issues such as the need for flood protection, aquatic ecosystem/water protection, nature/endangered species conservation, land use planning or conversion, rural development, or agricultural management. Meeting the needs of environmental legislation (Table 1) has pushed managers and policy-makers to demand landscape-scale products. Such products include estimates of habitat capacity for endangered species across vast areas of the Pacific Northwest (PNW) USA (e.g., Carmichael and Taylor, 2010; Burnett *et al.*, 2007), modeled estimates of river condition across the European Union (EU) (e.g., Pont *et al.*, 2006), or rigorous assessments of water body status and trends across the western USA (e.g., Pont *et al.*, 2009; Stoddard *et al.*, 2005) and the conterminous USA (e.g., USEPA, 2009; Paulsen *et al.*, 2008). Large-scale, multi-million dollar rehabilitation projects such as those being implemented in the U.S. for Chesapeake Bay, San Francisco Bay, south Florida, or entire states (e.g., Ohio and Oregon) require landscape-scale monitoring and analyses (Independent Multidisciplinary Science Team, 2007). Conforming to these directives and monitoring the effectiveness of large-scale restoration has created a "window of opportunity" for increased research at the landscape scale and for landscape perspectives to affect on-the-ground management (Hughes *et al.*, 2008).

The European Water Framework Directive (WFD), implemented in 2000, is an example of a relatively new policy directive that promotes landscape-scale analyses and management. It demands that all water bodies across Europe achieve good ecological status or good ecological

Table 1: Examples of policy and management drivers of landscape-scale approaches with region or country in parentheses (modified from Hughes *et al.*, 2008). See Section 2.1 for descriptions of policies.

Forces for change	Management needs	Law/Directives	Examples of landscape-scale response
Floods, climate change	Flood protection, water storage	Flood Protection Directive (Europe); The River Law (Japan)	Prioritization of areas for flood retention, flood risk management.
Degraded aquatic ecosystems including altered hydromorphology and decreased water availability	Aquatic ecosystem/water protection and storage	Water Framework Directive (Europe), Clean Water Act (USA), Endangered Species Act (USA), Manuelzão Project (Brazil), Environment Protection & Biodiversity Conservation Act (Australia), Species at Risk Act (Canada), Constitutional Rights of Nature (Ecuador), Ramsar Convention on Wetlands (Iran)	Protection and enhancement of the status of aquatic ecosystems and, with regard to their water needs, terrestrial ecosystems and wetlands directly depending on the aquatic ecosystems.
Decreasing biodiversity / increasing number of threatened/extinct species/habitats, increased numbers of invasive species	Nature conservation	Habitat and Birds Directive (Europe), Endangered Species Act (USA), Manuelzão Project (Brazil), Environment Protection & Biodiversity Conservation Act (Australia), Species at Risk Act (Canada), Constitutional Rights of Nature (Ecuador)	Political commitment to halt biodiversity loss within the EU by 2010. Establishment of a coherent network of protected areas. Guarantee ecological requirements of the natural habitat types and species of community interest. Nature protection.
Linkage of flooding events to land use	Land use planning, discourage floodplain occupation by humans	European Spatial Development Perspective (Europe)	Concentrating/clustering land use types according to their flood tolerance. Maintaining and re-activating natural flood retention areas.
Spatial disparities	Rural development	EU Rural Development Regulations (Europe)	Integrated approaches to sustainable rural development.
Health concerns, food supply, environmental degradation, ecosystem services	Agriculture, fisheries	Reformed common agricultural policy Agenda 2000 (Europe); Farm Bill (USA)	A long-term perspective for sustainable agriculture and fisheries; riparian/stream protection

potential by the year 2015 or, at the latest extension of the deadline, by 2027 (Water Framework Directive, 2000). These assessments are required for every stream and river in Europe and can be based on empirical or modeled data. The WFD specifically addresses impacts of catchment-scale land use on, for example, diffuse pollution in riverine systems. "Land use patterns, including identification of the main urban, industrial and agricultural areas and, where relevant, fisheries and forests" must be estimated (WFD Article 5 and Annex II) during the identification of human impacts in each catchment. Additionally, River Basin Management Plans (RBMP), required for all catchments, should include an "estimation of diffuse source pollution, including a summary of land use" (WFD, Article 13 and Annex VII). Because of these specific requirements, the WFD institutionalizes a large-scale approach to river research and management that requires coordination of activities such as monitoring, rehabilitation prioritization, planning, and public consultation across vast extents (Steyaert and Ollivier, 2007; Moss, 2004).

The WFD is not the only policy that promotes landscape-scale research and management across Europe. The Habitats Directive (1992) and the Birds Directive (1979) mandate a landscape approach to conservation planning that goes well beyond the traditional research scales of individual habitats and river reaches, even beyond political boundaries and frontiers. Such large-scale thinking is also reflected in the International Commission for the Protection of the Danube River (ICPDR), which focuses on the management of land including the built environment. The Natura 2000 network of conservation sites is yet another European example of a very large-scale conservation program and new policies continue to expand its scope.

There are policy drivers for landscape-scale analyses in other regions. One of the primary drivers of landscape-scale riverine research in the USA has been the Endangered Species Act (ESA; Public Law (PL) 92-205, 16 United States Code (USC) 1531 et seq. 1973). The ESA provides for the conservation of ecosystems upon which threatened and endangered species of fish, wildlife, and plants depend. Under this Act, twenty-seven evolutionarily significant units (ESUs) of anadromous Pacific salmonids, including Chinook, chum, coho, sockeye, and steelhead trout, have been listed as threatened or endangered since 1991. The cumulative area (spanning portions of the states of Washington, Oregon, Idaho, and California) of ESUs with one or more listed species is approximately 372,000 km^2 and encompasses thousands of kilometers of stream. To complicate matters, these fish use a wide variety of stream habitat types often spawning in low gradient habitats, rearing in smaller streams, and migrating through urbanized mainstem rivers. As a result of the ESA and the listing of Pacific salmonids over such vast extents, a critical need has arisen for estimation and prediction of species distributions and environmental conditions over large areas for which adequate field data do not exist. The Species at Risk Act adopted in Canada in 2002 serves a purpose similar to the Endangered Species Act and has promoted large-scale thinking about aquatic systems across Canada (`http://laws.justice.gc.ca/eng/S-15.3/`).

A second key policy driver across the USA is the Federal Water Pollution Control Act, commonly referred to as the Clean Water Act (CWA, PL 92-500, 33 USC 2101 et seq. 1948). The purpose of the CWA is to restore and maintain the chemical, physical, and biological integrity of the nation's waters. As a result, the US Environmental Protection Agency (USEPA) is involved in a wide range of monitoring and analysis programs that consider impacts of land use over large spatial extents on the health and condition of surface waters (lakes, streams, rivers, estuaries, coastal waters, wetlands). These programs are designed to answer questions such as: What is the current extent of the waters that support healthy ecosystems? Is water quality improving? What are the key stressors that account for waters with poor biological condition? The evolution of CWA programs over the last decade has included a shift toward watershed-based TMDL strategies (Total Maximum Daily Loads; `http://www.epa.gov/watertrain/cwa/`) and rotating probabilistic surveys of all USA surface waters on an annual basis (e.g., USEPA, 2009; Paulsen *et al.*, 2008; Shapiro *et al.*, 2008).

Japan is experiencing a similar shift toward riverine policy directives that promote a landscape-

scale perspective. The River Law in Japan was originally designed solely for flood protection and water use but, in 1990, Japan launched "Nature-Oriented River Work" or "Ta Shizen Gata Kawa Zukuri," which is an initiative aimed at conserving beautiful landscapes and riverine biodiversity. The trend toward increasing spatial extents in river conservation continued when the River Law was amended in 1997 to add "Conservation and Improvement of River Environments" as a third purpose of the law. These policies are expected to serve as a comprehensive and systematic plan to protect biodiversity and rehabilitate fluvial environments and ecosystems. One practical outcome of this strategy is the National Census on River Environment, which provides periodic surveys of the status of rivers and dams from an environmental perspective (e.g., fish survey) over large areas.

Beyond Europe, North America, and Japan, the trend toward large-scale conservation is also strong. The Ramsar Convention on Wetlands, signed in Iran in 1971 promotes a landscape perspective. In one of the latest resolutions (Resolution IX.I, Annex C) of this convention, it is stated that "management and development of wetlands must be undertaken within the context of their larger surrounding 'waterscape' (the river basin or catchment, including the hydrological processes and functions within the basin or catchment) as well [as] their larger surrounding landscape" (`http://www.ramsar.org/pdf/lib/lib_handbooks2006_e07.pdf`). Similar conditions led to the initiation of the River Health Program (RHP) in South Africa in 1994. The RHP objectives are to monitor and report on aquatic ecosystem status, trends, and emerging problems of South African rivers (Kleynhans, 1999; Murray, 1999). In Australia, over-allocation of water coupled with a long-term shift to a drought regime resulting from climate change have stimulated a Sustainable Rivers Audit Program to monitor conditions across the nation's major river basins (Davies *et al.*, 2006). In Brazil, the Manuelzão Project is directed toward public education, wastewater treatment, land and water use, and ecological monitoring across the entire Rio das Velhas Basin (Lisboa *et al.*, 2008). Brazil has also recently passed progressive environmental protection laws designed to protect the nation's rich natural resources, especially its water quality and biodiversity, and has established national parks to protect biodiversity and the recovery of large tracts of riparian corridors in many areas (Drummond and Barros-Platiau, 2006). Ecuador recently adopted a new constitution that formally recognizes the rights of nature "to exist, persist, maintain and regenerate its vital cycles, structure, functions and its processes in evolution" (`http://celdf.org/article.php?id=712`).

2.2   Ecological drivers

The nature of current conservation concerns has clearly necessitated a landscape-scale approach to a wide range of riverine issues (Table 1). A summary of all current conservation concerns is beyond the scope of this review; we simply remind readers of the large spatial extent of current issues. Managing watersheds, stream habitats, and fisheries in the face of climate change will, for example, require spatially extensive approaches to monitoring, data collection and analysis. Non-indigenous species continue to pose major threats to aquatic ecosystems (Leprieur *et al.*, 2008b; Lomnicky *et al.*, 2007; Miller Reed and Czech, 2005). Aquatic non-indigenous species range over very large areas and efforts to limit their spread or estimate their impacts must necessarily incorporate data collection and analysis techniques at the landscape scale (e.g., Sanderson *et al.*, 2009). Increasing awareness of and concern about these threats provides a second "window of opportunity" for comprehensive, large-scale river research (Hughes *et al.*, 2008).

2.3   Technological drivers

The incorporation of landscape thinking into riverine research has been catapulted by the availability of new technologies (Johnson and Host, 2010; Johnson and Gage, 1997). Habitat assessments in the past were necessarily transect-based and field intensive. Research and analyses had to proceed by sampling discrete reaches of very large, diverse, and dynamic systems. Therefore, despite

conceptual models that placed streams in their landscape context (e.g., Schlosser, 1991; Vannote *et al.*, 1980; Hynes, 1975), aquatic research methods have historically been focused on individual stream reaches.

Aerial photography, in use since the 1940s, was the first technology to provide a landscape-scale perspective but a suite of recent new technologies has dramatically improved our ability to conduct research over large areas. High spatial resolution (e.g., 1-meter pixel resolution) digital imagery provides powerful new tools for riverine landscape mapping and feature identification. Remote sensing has previously enabled synoptic views of entire rivers and their catchments and can now provide data at resolutions that capture reach-scale riparian and instream habitat structuring (Hall *et al.*, 2009; Silva *et al.*, 2008; Legleiter, 2003; Sawaya *et al.*, 2003; Leuven *et al.*, 2002; Marcus, 2002). Geographic Information Systems (GIS) have made landscape pattern analysis possible. As well, digital terrain modeling has provided tools for defining catchment boundaries and stream flow in flexible and easily repeatable ways.

Spatial data can describe relatively immutable natural conditions such as elevation, geology, soil type, air temperature, and precipitation. Spatial data are also available to describe anthropogenic stressors such as road density, land use, urbanization, migration barriers, water withdrawals, dams, and mine claim density. Mertes (2002) provided a comprehensive review of remote sensing advances and their applications to riverine environments, and discussed the newly arising capabilities to examine "spatial and temporal relationships among biota, hydrology, and geomorphology across scales from microhabitats to channel units to valleys to catchments". Mertes (2002) surveyed the increasing variety of sensing technologies available for mapping both landscape and water properties including optical sensors, light detection and ranging (LiDAR), forward looking infrared (FLIR), and radio detection and ranging (RADAR), deployed from multiple platforms from low altitude tethered balloons, to helicopter and fixed wing aircraft, to satellites. These new techniques allow us to capture information with a high degree of both lateral and longitudinal resolution. It is now also possible to directly sense water properties including elevation of the water surface, width of flooded area, distribution of wood, surface temperature, turbidity, and sediment movement (Smikrud *et al.*, 2008; Smikrud and Prakash, 2006; Mertes, 2002). And, we can generate stream network maps at much finer scales using LiDAR-based digital elevation models (DEM) than with satellite generated data (Mouton, 2005) as well as bathymetric delineations (Jones *et al.*, 2008; McKean *et al.*, 2008).

In addition to advances in spatial resolution, new sensors have increased spectral resolution. "Hyperspectral" sensors image in hundreds of finely defined wavelength bands, in contrast to the 3–6 broadly defined wavelength bands of color aerial photography and multispectral land imaging satellites (e.g., Landsat, SPOT). Legleiter *et al.* (2002) and Marcus (2002) compared capabilities of multi- and hyperspectral sensors for mapping instream features and both found that higher spectral and spatial resolutions increased mapping accuracies over lower resolution data. One outcome of these studies was the interesting prospect that mapping from high-resolution hyperspectral imagery might be more accurate than field based mapping (Marcus, 2002). Datasets generated from hyperspectral imagery have more stringent requirements for accurate preprocessing and require increasingly sophisticated classification techniques (Aspinall *et al.*, 2002; Wulder *et al.*, 2004); however, the spatial and spectral resolution provided by high-resolution hyperspectral systems is opening new avenues for ecosystem mapping, including mapping aquatic habitat features in both large and small streams (Figure 2). Traditional reach-based models, for example, have been limited in their ability to represent lateral habitat features but, using these new techniques, we can now map floodplains and large rivers accurately across two and even three dimensions (Kinzel, 2009).
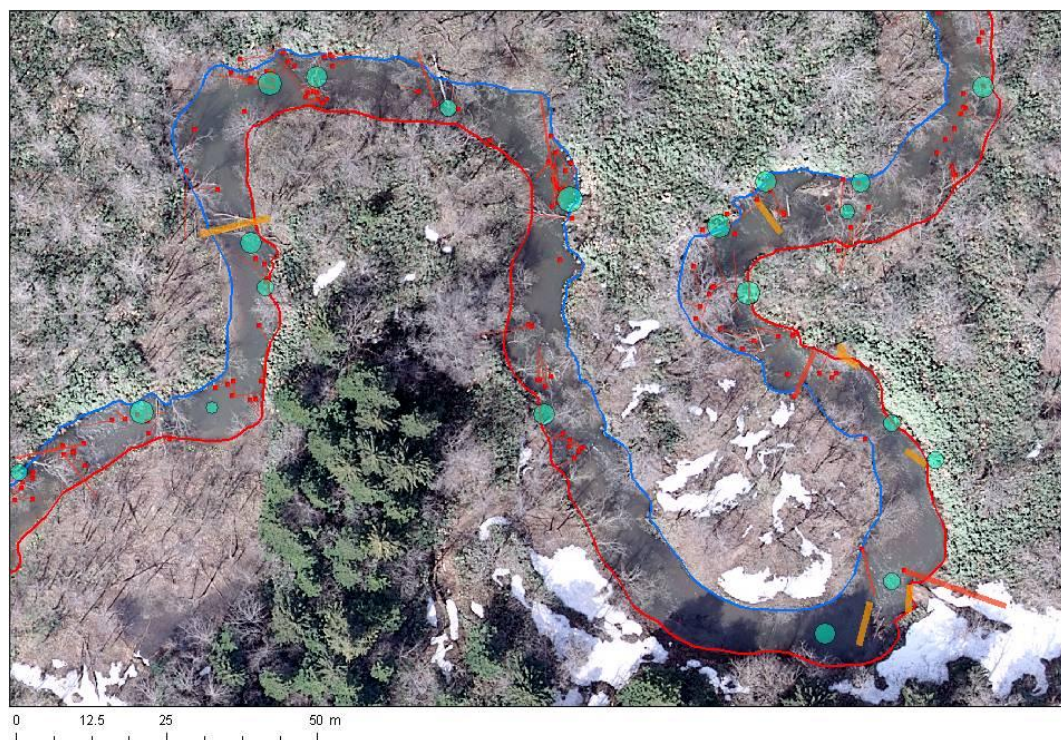
Figure 2: Aerial photo of a tributary of the Sarufutsu River, Japan, by DMC (digital mapping camera). Field-validated instream features such as woody debris (red lines), log jam (thick orange line), and deep pools (green circles) are superimposed. The field mapping of these features closely matches the photo imagery.

3   Conceptual history of landscape-scale riverine research

A series of landmark papers has developed the conceptual framework for landscape-scale riverine research. Hynes (1975) described the complex interactions linking terrestrial and aquatic ecosystems. Vannote *et al.* (1980) provided a coherent framework from which to predict and understand how interactions between terrestrial and aquatic ecosystems control instream processes and how these interactions change in a predictable way from forested headwater streams to mainstem rivers. Frissell *et al.* (1986) described how climate, geology, and land form provide a series of nested controls through which instream habitats are shaped and maintained, and to which biota respond. Ward (1989) described the four-dimensional nature of river ecosystems and the importance to aquatic biota and processes of maintaining those connections across landscapes and through time. Tonn (1990) offered a conceptual framework outlining how continental, regional, water body type, and local environmental processes and features (filters) govern fish assemblage patterns across all those spatial extents. Schlosser (1991) emphasized the importance of spatial and temporal heterogeneity and connectivity to river ecosystems and their biota. And, Hawkins *et al.* (1993) promoted the concept of hierarchical drivers of stream habitat features. Poff (1997) added a framework for using explicit multi-scale landscape constraints to predict and understand distribution, abundance, and local assemblage composition. Ward (1998) described the pathways by which landscape disturbances organize and constrain instream habitats. The next conceptual leap came from Fausch *et al.* (2002), who promoted the need for a continuous view of rivers and encouraged researchers and managers not to view the river as a series of disjoint reaches but to consider "the entire spatially heterogeneous scene of the river environment, the *riverscape,* unfolding through time." Burcher *et al.* (2007) applied the concept of an ecological cascade or series of impacts to understand how land-cover change can alter biological responses. Building on earlier ideas in which processes of higher scales form boundary conditions for lower scales (Angermeier and Winston, 1999), they proposed the land cover cascade as a conceptual framework for organizing the diverse relationships between landscape-scale phenomena and instream biological responses.

These theoretical frameworks have supported a large body of research to understand and to quantify interactions between landscape conditions over large spatial extents and instream responses. A series of review papers and research compendia organizes and tracks growth in the field of landscape riverine research.

A special issue of *Freshwater Biology* (1997) first began to assemble the ideas and findings of this new perspective on river systems. Johnson and Gage (1997) reviewed the state of the art at the time. They concluded that the advent of GIS technologies and the need for wide-ranging policies and management programs had pushed landscape-scale research to the forefront of riverine ecology. They synthesized the landscape metrics in common use for landscape analysis and the statistical tools available for linking landscape patterns to instream conditions. Johnson and Gage concluded that these new spatial and statistical tools have enabled ecologists to study patterns and relationships over larger and more diverse extents than previously possible. Other papers in the special issue compared local versus landscape perspectives on Michigan trout streams (Wiley *et al.*, 1997), examined the role of catchment land use patterns on stream integrity (Allan and Johnson, 1997), land use on macroinvertebrate assemblages in New Zealand (Townsend *et al.*, 1997), landscape condition on water chemistry (Johnson and Gage, 1997), and multi-scale impacts on species traits of macroinvertebrate assemblages (Richards *et al.*, 1997).

A second set of papers was assembled in *Freshwater Biology* in response to the *First International Symposium on Riverine Landscapes.* In that special issue, Wiens (2002) promoted the direct application of concepts from the field of landscape ecology, which had traditionally focused on terrestrial landscapes, to aquatic systems. Papers in the special issue described linkages between landscapes and both hydrogeomorphic processes (Poole, 2002) and biological responses (Malmqvist, 2002), detailed advances in remote sensing and GIS technologies (Mertes, 2002) and

applied landscape concepts to riverine management (Poudevigne *et al.*, 2002).

A comprehensive and detailed review of research linking land use to instream conditions was recently provided by Allan (2004a). He outlined pathways by which land use at the catchment and riparian scales affects instream habitats (Table 2) and synthesized over 100 research papers correlating land use with metrics of stream health. Allan found that the most common landscape composition metric in studies linking land use and land cover to aquatic response has been the percentage of land cover types in either the catchment or the riparian area. Across multiple projects, he was able to conclude that a high proportion of forest cover in a catchment is normally associated with positive stream conditions. Conversely, agricultural or urban areas in the catchment have been documented to have a negative influence on downstream river conditions (Allan, 2004a). Allan (2004b) also raised several key challenges to linking landscapes and the ecological status of rivers.

Durance *et al.* (2006) reviewed 658 papers dealing with the management of fish in rivers and found that only 27 explicitly considered the impact of scale. They were able to summarize some catchment-scale pathways by which landscapes affect fish abundance and distribution such as climate, geology, vegetation, and past land use (Table 3) (Durance *et al.*, 2006).

In 2006, a large volume of research exploring and quantifying landscape influences on stream habitats and biological assemblages was assembled by Hughes *et al.* (2006b). This collection of 30 research papers focused on research linking catchment-scale landscape condition to instream habitat and fish responses. The editors concluded that there were four key challenges in studying river systems in a landscape context: determining appropriate units for measuring and interpreting the riverscape, understanding the mechanisms by which land use alters river habitats and biota, measuring and understanding how spatial factors interactively affect aquatic habitats and biota, and collecting and interpreting appropriate landscape and riverine data. Major knowledge gaps requiring additional research included improving river-landscape classification, determining the appropriate spatial and temporal scales at which data should be captured, improving predictive models where data are limited, and improving our measures of connectivity among river networks and their landscapes.

Most recently, Johnson and Host (2010) reviewed the landscape-aquatic literature from 1986 to 2008, revealing a 20-fold increase in the number of publications over that time. They also found that as the scales of the studies increased so did the importance of landscape versus site-scale predictors. Because of the use of varied terms and extents for 'site' and 'landscape', they recommended that editors demand explicit descriptions of both. In the same issue, Poole (2010) pointed out that stream hydrogeomorphology is also at the roots of riverine landscape ecology by providing mechanistic foundations for how fluvial landscapes shape stream ecosystems.

Table 2: Principal mechanisms by which land use influences stream ecosystems (modified from Allan, 2004a).

Environmental factor	Effects
Sedimentation	Increases turbidity, scouring and abrasion; impairs substrate suitability for periphyton and biofilm production; decreases primary production and food quality causing bottom-up effects through food webs; in-filling of interstitial habitat harms crevice-occupying invertebrates and gravel-spawning fishes; coats gills and respiratory surfaces; reduces stream depth heterogeneity, leading to decrease in pool species.
Nutrient enrichment	Increases autotrophic biomass and production, resulting in changes to assemblage composition, including proliferation of filamentous algae, particularly if light also increases, accelerates litter breakdown rates and may cause decrease in dissolved oxygen and shift from sensitive species to more tolerant, often non-native species.
Toxic chemicals	Increases heavy metals, synthetics, and toxic organics in suspension associated with sediments and in tissues; increases deformities; increases mortality rates and alters abundance, drift, and emergence of invertebrates; depresses growth, reproduction, condition, and survival among fishes; disrupts endocrine system; physical avoidance.
Hydrologic alteration	Alters runoff-evapotranspiration balance causing increases in flood magnitude and frequency, and often lowers base flow; contributes to altered channel dynamics, including increased erosion from channel and surroundings and less-frequent overbank flooding; runoff more efficiently transports nutrients, sediments, and contaminants, thus further degrading in-stream habitat. Strong effects from impervious surfaces and stormwater conveyance in urban catchments and from drainage systems and soil compaction in agricultural catchments. Dams can alter flow patterns, removing natural fluctuations that produce channel complexity and/or producing sudden unnatural fluctuations.
Riparian clearing / canopy opening	Reduces shading, causing increases in stream temperatures, light penetration, and plant growth; decreases bank stability, inputs of litter and wood, and retention of nutrients and contaminants; reduces sediment trapping and increases bank and channel erosion; alters quantity and character of dissolved organic carbon reaching streams; lowers retention of benthic organic matter owing to loss of direct input and retention structures; alters trophic structure.
Loss of large woody debris	Reduces substrate for feeding, attachment, and cover; causes loss of sediment and organic material storage, reduces energy dissipation; alters flow hydraulics and therefore distribution of habitats; reduces bank stability; influences invertebrate and fish diversity and community function.

Table 3: Environmental variables associated with fish distribution, classified by system attribute and observational scale. Synthesis is based on 658 papers dealing with management of fish in rivers, including 27 papers that acknowledge and compare across scales (modified from Durance *et al.*, 2006).

System attribute	Scale of environmental variable		
	Catchment	*Segment/reach*	*Habitat/riparian*
Temperature variability	Climate, elevation, drainage area	Channel morphology	Vertical hydraulic exchanges
Hydrologic regime	Climate, geology and drainage area	Channel morphology/complexity, reach size, reach elevation	Habitat morphometry/complexity and depth
Spatial configuration	Connectivity with other waters	Lateral/catchment connectivity, dams	Connection with main stream, log weirs
Chemistry	Water chemistry	Water chemistry	Water chemistry
Biotic features	Catchment vegetation, biome, land use	Cover, land use	Food resources, predator/competition, riparian vegetation/land use
History	Past biome, climates, land use, dams	Past land uses, temperatures, dams	Dams

4 Challenges to landscape riverine research

4.1 Has new research utilized the strengths of new technologies or are we doing the same old stuff with more expensive data?

The vast quantity of readily available spatial data sets over large areas, new methods for collecting spatial data with smaller and smaller grain sizes, and the improved computing power of spatial analytic software suggest that whole new approaches to landscape riverine ecology are now possible. By providing synoptic views of streams and their catchments at multiple resolutions, and by providing the spatial analytical framework to exploit the power of digital maps, these new technological tools provide the capability to examine interrelationships between streams and their catchments in ways that were either extremely limited or impossible in the past. Our first question is whether new research is truly harnessing the power of these new resources or whether we are simply doing the same old things with new data that are more resource-intensive to collect.

Mapping the water itself

Habitat parameters particularly amenable to remote sensing include water temperature and stream depth; substrate type can also be collected under certain conditions. Analyses that apply these new types of data have made considerable progress in our understanding of how streams and landscapes interact. The work of Torgersen *et al.* (1999, 2001), for example, demonstrated how high resolution airborne FLIR systems could be used to assess the spatial distribution of thermal habitats relevant to stream fishes of the Pacific Northwest, USA. Techniques developed by Fonstad and Marcus (2005) link spectral reflectance, stream discharge measured at local gages, and equations describing stream resistance into predictive models of depth termed "hydraulically assisted bathymetry". These techniques and similar efforts by Bjerklie *et al.* (2003, 2005) hold promise for mapping stream depth over entire stream networks without the need for field data. Additionally, retrospective studies using archival photography have examined temporal trends in stream depth due to land use changes (Fonstad and Marcus, 2005). In a separate study, Lorang *et al.* (2005) linked field-derived estimates of water depth, flow velocity, shear stress, and stream power to multispectral imagery to spatially model geomorphic processes in gravel-bed rivers. While these techniques are promising, Legleiter and Roberts (2005) found that channel morphology could affect the accuracy of image-derived depth estimates.

Employing new instream habitat data

Recent years have seen remarkable advances in application of high resolution remote sensing to the study of stream physical habitat structure. Newly available systems enable pixel resolutions of 1-meter or better, providing the fine spatial resolution needed to detect and map instream features of small rivers and streams. Wright *et al.* (2000) used 1-meter resolution multispectral (e.g., 4 wavelength band) digital imagery to map stream morphological units (eddies, glides, riffles, scour pools, etc.) of 3rd and 4th order streams with moderate to high levels of success. Then, in a series of pioneering studies (Legleiter *et al.*, 2004; Legleiter, 2003; Marcus *et al.*, 2003), high spatial resolution hyperspectral imagery was shown to be effective at mapping instream habitat features including woody debris, depth, and substrate. The authors tested spectral band ratios and classification methods for extracting stream habitat information from spectral imagery (Legleiter, 2003; Marcus *et al.*, 2003) and developed the basis for physical models relating spectral reflectance to instream features (Legleiter *et al.*, 2004). Their results demonstrate the feasibility of these technologies for mapping small streams, although higher accuracies were possible in larger streams (Marcus *et al.*, 2003). Leckie *et al.* (2005) exploited high spatial resolution (80 cm) multispectral data for mapping instream habitat and were able to reliably map substrate, woody debris, and

depth classes. Feurer *et al.* (2008) reviewed progress in mapping underwater topography using radiometric models and through-water photogrammetry. McKean *et al.* (2009a,b, 2008); Madriñán (2008); Kinzel *et al.* (2007) have had success mapping stream and riparian habitat using LiDAR.

In addition to physical habitat features, direct sensing of submerged aquatic vegetation and algae are now possible by combining high resolution imagery and GIS modeling. Lehmann and Lachavanne (1997) provided a comprehensive review of the topic, although at the time most applications were limited to larger rivers or estuaries, primarily because of a lack of data at the appropriate scale. The interaction of light with the water column introduces spectral classification and interpretation issues unique to aquatic environments, but the theoretical basis for remote sensing of aquatic vegetation has been established (Silva *et al.*, 2008). Although applications to small streams are still rare, the potential of hyperspectral imagery for mapping aquatic vegetation over entire catchments is evident (Govender *et al.*, 2007; Nelson *et al.*, 2006).

Impediments and opportunities to harnessing new technologies

Most aquatic mapping applications to date have used passive optical remote sensing to relate stream properties to spectral reflectance patterns in imagery. Interpreting and extracting information from these data requires a high level of operator skill and data acquisition over large areas is expensive. Marcus and Fonstad (2007) reviewed applications to date and discussed the potential and limitations of this technology for advancing river science. While they predicted that future developments of optical remote sensing would create continuous meter-scale maps of stream habitats across entire river systems, they rightly discussed obstacles and issues that must be addressed to insure future advancement. These problems include difficulty mapping areas with steep terrain (valleys and canyons), differential illumination of stream habitats because of shading and surface turbulence, the need for clear water conditions for assessing instream habitat features and vegetation, timing constraints in image acquisition because of satellite overpass or weather conditions, the cost of acquiring repeat imagery, difficulties assessing mapping accuracy of stream features, and ethical issues related to revealing habitat locations that could be easily exploited by anglers (Marcus and Fonstad, 2007). As well, promising new data are often available for ecological research only in limited pilot areas and there is a disproportionate amount of data for large versus small rivers. Some parameters are still limited to clear waters or surface waters (e.g., temperature). It often remains prohibitively costly to assess dynamic stream habitat environments from repeated flights of static airborne imagery. The increased availability of remote sensing data, as well as field data collected from different large-scale monitoring programs, creates substantial data aggregation problems (Roper *et al.*, 2010; Independent Multidisciplinary Science Team, 2009) and the costs of obtaining, storing, manipulating, and interpreting such imagery are nontrivial.

In spite of these limitations, great potential exists for further refinement of optical remote sensing techniques, especially when combined with other sensing technologies to assess the complete aquatic environment (Marcus and Fonstad, 2007; Leckie *et al.*, 2005). While just beginning, the application of high resolution active remote sensing methods such as LiDAR for mapping stream morphology and habitat features (McKean *et al.*, 2009a,b, 2008; Feurer *et al.*, 2008; Kinzel *et al.*, 2007) holds promise for precise mapping of channel morphology and water surface elevation. When combined with optical methods (Hall *et al.*, 2009) and thermal imaging, the possibility exists for mapping multiple stream parameters over large regions for a nearly complete assessment of fish habitats. Host *et al.* (2005), for example, published a complex spatial analysis that provided an efficient method for the identification of reference areas along the Great Lakes coast, USA.

All relevant technological advances will not be solely in the collection and processing of spatial data. In ichthyological and fisheries sciences, researchers have recently developed a technique to reconstruct migration history of individual fishes based on chemical analysis of bony tissue, otoliths. Otolith microchemistry, especially when combined with landscape-scale data, could provide new

avenues to study fish migration across large areas and to identify effects of landscape disturbances such as dams on riverine fishes (e.g., Clarke *et al.*, 2007; Hogan and Walbridge, 2007).

4.2   Have we incorporated key concepts from landscape ecology to improve our understanding of how landscapes affect rivers?

The field of landscape ecology has focused on the relationships between pattern and process as well as on planning for patterns of human land use (Forman and Godron, 1986). Within landscape ecology, rivers have traditionally been characterized as landscape elements (Wiens, 2002) rather than as the target of large-scale processes or as the research focus. There have been many calls to apply principles of landscape ecology in river ecology (e.g., Wang *et al.*, 2006a; Wiens, 2002) or to incorporate the dynamic elements of rivers into landscape thinking. Below, we briefly describe five elemental concepts from landscape ecology and how they have been applied to improve our understanding of how landscapes alter rivers.

Relationships between Patterns and Processes

A fundamental underpinning of the field of landscape ecology is the study of relationships between patterns and processes, usually over broad spatio-temporal scales (Turner, 2005). Specifically, landscape ecologists seek to understand how ecosystem processes act to form patterns on the landscape, and of equal interest, how landscape patterns can influence ecological processes. Aquatic ecologists have made great strides toward understanding ecosystems by including this important concept (Fausch *et al.*, 2002; Ward *et al.*, 2002; Townsend, 1996). For example, researchers have investigated how ecosystem processes such as climate, hydrology, and geomorphology influence stream characteristics (e.g., Benda *et al.*, 2004; Swanson *et al.*, 1988) and fish distribution patterns (e.g., Pess *et al.*, 2002). More prevalent are examples of the possible effect of upland landscape composition on the functioning of instream processes. Researchers have related distributions of natural features in the surrounding landscape, such as topography and soils (e.g., Richards *et al.*, 1996), and human land use (e.g., Independent Multidisciplinary Science Team, 2010; Brown *et al.*, 2005; Kershner *et al.*, 2004; Van Sickle *et al.*, 2004; Snyder *et al.*, 2003; Paul and Meyer, 2001; Roth *et al.*, 1996) to instream biodiversity and viability of fishes.

To date, most landscape-scale research on lotic systems has attempted to evaluate relationships between pattern and process by quantifying the effect of the composition of surrounding upland landscapes (e.g., 40% of a watershed is in low-density residential land use) on some instream feature (e.g., fish abundance Paulsen and Fisher, 2001). Inclusion of metrics of landscape structure in both upland landscapes (e.g., connectivity of non-impervious areas or average patch size of high quality riparian forest; Gergel *et al.*, 2002) and within aquatic landscapes (e.g., longitudinal connectivity of suitable habitat patches; Isaak *et al.*, 2007; Torgersen *et al.*, 2006; Benda *et al.*, 2004; Wiens, 2002) is an opportunity to clarify our understanding of the interaction between terrestrial and aquatic systems.

Scale

Integral to both landscape ecology and aquatic ecology is recognition that relationships between organisms and their habitats depend on the spatio-temporal scales at which they are observed (e.g., Talley, 2007; Durance *et al.*, 2006) (Figure 1). Scale is clearly important in identifying and analyzing spatial pattern (Feist *et al.*, 2010; Wu, 2004; Wu *et al.*, 2002; Turner *et al.*, 1989) and has garnered much attention over the past 20 years or so (Levin, 1992; Wiens, 1989). Schneider (2001) dedicated a special section to addressing the many ways that scale can be defined and Jenerette and Wu (2000) wrote an entire essay on the multiple definitions of scale. The emphasis of landscape ecology on scale has led riverine landscape research to use multi-scale studies in order

to gain a better understanding of processes acting in a stream network (Lowe *et al.*, 2006; Leclerc and DesGranges, 2005; Fausch *et al.*, 2002; Lammert and Allan, 1999; see Table 3 in Johnson and Host, 2010, for further examples). Because of the strong migratory behaviour of many fishes, they are ideal study subjects for testing hypotheses about scale (Schmutz and Jungwirth, 2001). Processes affecting fish assemblages range from global to local scales (Durance *et al.*, 2006; Tonn, 1990).

Existing cross-scale studies have provided contradictory results giving either more weight to local (Walters *et al.*, 2003) or catchment factors (Mugodo *et al.*, 2006; Marsh-Matthews and Matthews, 2000). Studies that examine instream response to land use at multiple scales report, unsurprisingly, mixed influence (Feist *et al.*, 2010; Stewart, 2001; Fitzpatrick *et al.*, 2001; Richards *et al.*, 1996; Roth *et al.*, 1996; Johnson and Host, 2010; see Table 3 in Johnson and Host, 2010, for further examples). For example, near-stream connected imperviousness had a stronger influence on fish assemblages than did comparable amounts of impervious surface located farther from the stream, apparently owing to increased severity and frequency of high-flow events and lowered baseflow (Wang *et al.*, 2001); Yet, catchment-scale influence may be greatest when the primary mechanism is flow instability, nutrients, or some other factor related to the entire landscape (Allan, 2004a). Wang *et al.* (2006a) found that fish assemblages were mainly influenced by local factors in undisturbed catchments whereas the relevance of catchment scale factors increased with increasing landscape disturbance.

Opportunities for advancing the study of scale in landscape riverine research are emerging from the advances in technology described in Section 2.3, which are providing fine-grain data over large extents even on small rivers. To explicitly measure the influence of scaling on fish populations, we must hold the grain constant as we vary the extent, and vice-versa. There are many good examples in the literature of varying the analysis extent while holding the grain constant (e.g., Moerke and Lamberti, 2006; Creque *et al.*, 2005; Santoul *et al.*, 2005; Feist *et al.*, 2003). It is unusual to find riverine examples in which researchers varied the grain while holding the extent constant. There are examples from theoretical ecology, but these are usually in small systems under controlled conditions.

Connectivity

Landscape ecologists define connectivity as the degree to which the landscape facilitates or impedes the ability of organisms to move among resource patches (Taylor *et al.*, 1993). Hitt and Angermeier (2008b, 2006) found that failure to consider spatial connectivity may bias measures of biotic integrity. However, classic connectivity metrics used in two-dimensional ecosystems such as terrestrial or oceanic landscapes (Calabrese and Fagan, 2004) are not easily applied in stream networks for several reasons. First, quantifying connectivity in stream networks is more challenging than quantifying connections in two-dimensional habitats (Fagan, 2002). Second, resource patches in temporally dynamic aquatic systems typically experience a high degree of patch turnover (Gresswell *et al.*, 2006). Third, organisms must contend with the force of current velocity, which may affect locomotive abilities and alter directional movement among habitats (e.g., Olden, 2007; Gresswell *et al.*, 2006). Fourth, the apparent existence of different riverine zones that appear to support different fish assemblages suggests that fish species pools are filtered by various longitudinally varying river characteristics such that some species are limited to specific areas (Ibañez *et al.*, 2009; McGarvey and Hughes, 2008; Vannote *et al.*, 1980; Hawkes, 1975; Huet, 1949; Fritsch, 1872). Some recent progress has been made towards quantitatively measuring and studying connectivity in streams (Cote *et al.*, 2009; Hitt and Angermeier, 2008a; Hughes, 2007; Isaak *et al.*, 2007; Ganio *et al.*, 2005; Fagan *et al.*, 2002) but many opportunities to refine and explore the topic exist.

Landscape fragmentation by dams

Barriers and dams are a particularly pressing ecological problem (Graf, 1999). They fragment aquatic systems both by impeding fish migrations and by disconnecting physical processes such as the transport of wood, water, and sediment. In Europe, for example, dams and water pollution in the Rhine and Danube Rivers have reduced the native fish fauna, particularly salmonids and sturgeons (Bacalbasa-Dobrovici, 1989; Lelek, 1989). Likewise, dams have eliminated several diadromous fish species from the Seine River, France (Oberdorff and Hughes, 1992). And, a vast network of dams and flow alterations on the Colorado River and Rio Grande, USA, have resulted in deteriorated water quality, flows frequently failing to reach the sea, and replacement of endemic fish faunas by invasive non-indigenous species (Hughes *et al.*, 2005b). Among major USA rivers draining to the Pacific Ocean, those with mainstem dams without fish passage or with inadequate fish passage (e.g., Snake, Columbia, Klamath, Sacramento–San Joaquin Rivers) have all experienced substantial flow and channel alterations, water pollution, species extirpations, and reaches dominated by non-indigenous species (Brown *et al.*, 2005; Hughes *et al.*, 2005b; Ebel *et al.*, 1989). Even small low-head dams can eliminate native species and alter the distributions of non-indigenous species (LaVigne *et al.*, 2008a; Holden *et al.*, 2005; Meffe, 1984). The fish species most affected by instream anthropogenic barriers are species with a limited range of habitat types and migratory species. Understanding the magnitude of fragmentation caused by dams requires investigations over large extents.

One area where riverine landscape research has made strong progress is in quantifying the impacts of landscape fragmentation by dams on aquatic systems, and, in particular, on fish assemblages. Fukushima *et al.* (2007) were able to quantify impacts of watershed fragmentation on fish assemblages through use of spatially explicit models applied to the occurrence data of multiple freshwater fish species inhabiting Hokkaido Island, Japan (ca. 80,000 km^2). Sheer and Steel (2006) quantified a relationship between the amount and distribution of lost aquatic habitat as a result of dams or barriers and salmon population performance. Letcher *et al.* (2007) found that habitat fragmentation by barriers increased the likelihood of local and system-wide extinction. We did not identify research examples in which other human impacts, such as pollution, altered thermal regimes, or flow alterations, were explicitly considered through the lens of landscape fragmentation or reduced connectivity between habitats.

There are global inequities in our understanding of the impacts of river fragmentation on fish assemblages and communities. The scarcity of quantitative data in developing countries presents a tremendous challenge to assessing effects of human disturbances on riverine ecosystems. The lack of scientific data may be contributing to aggressive human development in these same regions. For example, the Mekong River, the 11th-longest river in the world, is being fragmented by a series of large dams (Baran *et al.*, 2007). Dam construction within the Mekong watershed is on the rise with increasing demand for hydroelectricity. The Yangtze River, the third longest river in the world, now has the world's largest dam, the Three-Gorges Dam, fragmenting an area of about 58,000 km^2 (Xie, 2003). These enormous hydrologic alterations are predicted to significantly reduce both terrestrial (Wu *et al.*, 2003) and aquatic (Park *et al.*, 2003; Xie, 2003) biodiversity, eventually diminishing fisheries resources and food security (Baran *et al.*, 2007). A "geowiki" is being developed by Mark Mulligan at King's College in London to help coordinate data-sharing and improve spatial coverage of data on dams (`http://www.kcl.ac.uk/schools/sspp/geography/research/emm/geodata/geowikis.html`). The database combines the visualization capabilities of Google Earth with the power of multiple Internet users to develop a comprehensive global database of dams.

Humans are integral to ecosystems

Economic and population growth dramatically influence the functioning of ecosystems. For example, economic prosperity has been associated with declines in biodiversity worldwide (Clausen and York, 2008; Leprieur *et al.*, 2008b; Naidoo and Adamowicz, 2001), and may be especially problematic for freshwater fishes (Miller Reed and Czech, 2005; Rose, 2005). Urbanization has significant impacts on freshwater ecosystems. The consistent hydrological effect of urbanization, including flashy flows (more frequent, short-lived, more intense), is a result of the increased amount of impervious surfaces in urban streams (e.g., Booth *et al.*, 2004). Researchers have also been able to detect more diffuse effects of urbanization and land use (e.g., Randhir and Ekness, 2009; Bilby and Mollot, 2008; Alberti *et al.*, 2007; Walsh *et al.*, 2005; Kershner *et al.*, 2004; Paulsen and Fisher, 2001; Paul and Meyer, 2001; Roth *et al.*, 1996). Human threats to imperiled fish populations also include the spread of non-native species (Leprieur *et al.*, 2008a) and global climate change (Crozier *et al.*, 2008; Rieman *et al.*, 2007; Flebbe *et al.*, 2006). A concept in landscape ecology that is becoming more evident in recent years is the notion that humans should be considered as part of the ecosystem being managed, rather than as an outside factor exerting negative impacts on natural systems (Otte *et al.*, 2007; Wu and Hobbs, 2002). Few riverine studies or approaches have incorporated this mindset.

4.3 Have we been able to use landscape analyses to address management and policy needs?

Results of landscape analyses have significant implications for management of streams and catchments as well as for developing informed policies on land management. The genesis of this field lies in great part with policy directives, the increasing need for catchment-scale management, and wide-ranging environmental problems as discussed in Section 2. Therefore, landscape analyses are often intended to have on-the-ground impacts. However, evaluating the degree to which landscape analyses have actually been used for making management decisions is difficult because many peer-reviewed publications define the possible or intended application rather than the actual application. Allan (2004a) noted that there are limitations in the degree to which these types of analyses can inform management prescriptions and fisheries management. He and others have indicated the need for a more experimental approach in which systematic variations in land use are considered. In this section, we identify the most common potential applications of landscape analyses for riverine management and those papers that explicitly focus on these applications.

Conservation of species, populations, and biodiversity

Because landscape ecology has traditionally involved evaluation of spatial patterns of land use and species distributions, it has followed naturally that newfound knowledge was applied to ecosystem management such as the creation of conservation reserve networks (Thieme *et al.*, 2007; Margules and Pressey, 2000). Landscape-scale riverine analyses often apply high spatial resolution satellite and aerial imagery to identify conservation and restoration needs of large and complex aquatic ecosystems. The Missouri Aquatic GAP Project, for example, used landscape-scale data to identify types of aquatic habitats not adequately represented within the existing conservation network (Sowa *et al.*, 2007). Randhir and Tsvetkova (2009) used landscape-scale data to explore conservation implications of relationships between water conservation and management strategies. And, Ballinger and Mac Nally (2006) used a landscape perspective to explore the impacts of spatial and temporal flooding variability on wildlife habitats in the Murray–Darling Basin, Australia. Statistical models were built and tested using species occurrence data for multiple fish species in the Iberian Peninsula (Filipe *et al.*, 2004). These models used landscape characteristics to quantify the multi-species conservation value of particular river reaches and, eventually, to select a series

of reserve reaches. Ekness and Randhir (2007) identified spatial criteria for watershed-scale policy development. They concluded that basing conservation networks on stream order, riparian condition, and land use could lead to increased riparian areas, protection of headwaters, and minimized disturbance in headwater areas. However Li *et al.* (1996) and Dunham *et al.* (2008) recommended protection of mainstem reaches and entire channel networks, and Osborne and Wiley (1992) and Hitt and Angermeier (2008b) have reported on the importance of confluences to fish species occurrences in tributaries.

Species-based examples of applying landscape-scale data to conservation management include restoration of the Middle Rio Grande in New Mexico, USA for the endangered silvery minnow (Cowley, 2006), estimation of potential steelhead habitat above barriers in the Willamette River basin, Oregon, USA (Steel *et al.*, 2004), or probability of occurrence of two diadromous fish species in New Zealand (Eikaas *et al.*, 2005). Landscape condition has also been used to estimate the frequency and severity of ecosystem threats, such as the frequency of human-induced stressors, to provide a more informed basis for conservation planning (Mattson and Angermeier, 2007; Kracker, 2006). An example of the use of large-scale analyses to identify specific landscape stressors and the impact of those stressors on fish assemblages is Scott (2006), who quantified the impacts of urbanization on endemic versus broad-ranging fishes.

Management of riparian areas and catchments

One of the most obvious and far-reaching impacts of landscape analyses for land use management has been the promotion of the riparian zone as a critical transition area between streams and their catchments. In rehabilitating lateral connectivity, studies have emphasized establishing or enlarging riparian wooded buffer zones (Miltner *et al.*, 2004; Freeman *et al.*, 2003; Snyder *et al.*, 2003; Wichert and Rapport, 1998). Riparian management is particularly attractive because of the riparian zone's immediate influence on stream condition and because it promises benefits that are highly disproportionate to the land area required (Allan, 2004a). The landscape perspective can improve management of riparian lands by identifying the best strategies for particular areas. For example, the landscape perspective enabled fine-tuning of forest buffer restoration priorities in agricultural areas of Maryland, USA by quantifying those underlying geologies for which forested buffers had the greatest impact on indices of biotic integrity (Barker *et al.*, 2006). FEMAT (1993) also took a landscape approach in its recommendation of a riparian buffer equivalent to 2 site-potential tree heights along fish bearing streams, because the size of trees varies by ecoregion. Multi-scale analyses have been used to demonstrate that riparian management shows greater effectiveness in protecting streams from the negative impacts of land use at the local or reach scale than at the catchment scale. Intensive land use at the scale of entire catchments may have impacts too great for a riparian zone to moderate (Wang *et al.*, 2006b; Allan, 2004a; Snyder *et al.*, 2003; Morley and Karr, 2002; Nerbonne and Vondracek, 2001; Roth *et al.*, 1996).

Use of landscape analyses to improve land management across entire catchments has been more challenging. Reversal of land use to a less-developed state over vast extents is usually infeasible; however, improvement of stream conditions can be accomplished by promoting best management practices (BMPs) and improvements in landscape management, e.g., reduced fertilizer applications in the catchment (Wang *et al.*, 2006c) or conservation tillage (Yoder *et al.*, 2005). However, where native fish species have been extirpated, improvements in land use alone may not be enough to restore species distributions to their historic states (Wang *et al.*, 2006c).

Bio-assessment

The landscape perspective has improved fish-based bio-assessments through validation of biotic indices and identification of appropriate sampling scales. Correlations between indices of biotic

integrity and landscape conditions have validated expected relationships between degraded landscapes and stream condition (e.g., Carlisle *et al.*, 2008; Pinto *et al.*, 2006; Bramblett *et al.*, 2005; Dauwalter and Jackson, 2004; Joy and Death, 2004). However linking instream biological responses to landscape (land use, economic growth, population growth; Schleiger, 2000; Leprieur *et al.*, 2008a) and riverscape (dams, diversions; Stanford and Ward, 2001; Ward and Stanford, 1995) stressors requires different analytical approaches than the site-specific studies that comprise most biological assessments. For example, Hawkins *et al.* (2000) found that while landscape data can explain more variation in aquatic biota than what might be expected by chance alone, the amount of variation explained is not large, and thus will have limited value in making reach scale predictions. Herlihy *et al.* (2006) reported that landscape classifications accounted for approximately half the variability in fish assemblage clusters of the conterminous USA; however Pinto *et al.* (2009) reported that ecoregion and fish species clusters had similar classification strengths at a river basin scale.

Many analyses correlating land use with indices of biotic integrity have compared models at varying spatial scales such as riparian buffer scales and catchment scales (e.g., Barker *et al.*, 2006; Van Sickle *et al.*, 2004). Such analyses have begun to identify the specific scales at which land use drives various types of biotic indices. Fish-based and macrophyte-based biological indices are often more closely correlated with ecological quality at the river basin scale whereas indices based on macroinvertebrates and benthic diatoms are more closely correlated with environmental metrics at the reach and stream scale (Springe *et al.*, 2006). Brazner *et al.* (2007) and Allen *et al.* (1999) reported similar differences among biological indicators of disturbance in northeastern USA lakes and Great Lakes wetlands, respectively. In addition, recent focus on stream network topology has demonstrated the importance of considering position in the stream network and dispersal rates when using bioindicators to assess fish assemblage responses to the environment (Hitt and Angermeier, 2008b). The landscape perspective inherent in these studies has led to improved use of biological indicators of stream health; however, the lack of specific and tested mechanistic relationships between remote landscape conditions and instream biological response continues to plague on-the-ground applications and to limit the applicability of results beyond the study basins.

Prioritization of rehabilitation activities

Another common application of the landscape perspective is in determining where, within a catchment, rehabilitation activities should be undertaken. Some approaches have used riparian condition to identify those sub-basins most likely to respond to rehabilitation activities (e.g., Fullerton *et al.*, 2006) others have identified locations within basins that are most likely to increase aquatic connectivity or provide cumulative positive impacts (Jansson *et al.*, 2007). Landscape characteristics have been used to estimate the quantity of lost habitat behind migration barriers (Sheer and Steel, 2006) and to predict the quality of habitat where fish currently do not exist, such as behind migration barriers, and prioritize barrier removal projects (Steel *et al.*, 2004).

A large-scale perspective is necessary for rehabilitation prioritization for three reasons. First, the watershed is the unit of observation. Abiotic conditions and biological assemblages within a watershed are interdependent and cannot be effectively examined in isolation. Second, funding is often distributed for political equity and/or by watershed and so decisions about what restoration actions to fund must prioritize across all possible actions within a political jurisdiction or watershed. And, third, rehabilitation actions are often aimed at reducing impacts associated with large scale degrading processes such as wood, sediment, nutrient, and water delivery (Lake *et al.*, 2007). Understanding these large-scale habitat-forming processes (Beechie and Bolton, 1999) across entire catchments aids us in identifying and prioritizing aquatic rehabilitation activities at the appropriate temporal and spatial scale. Weber *et al.* (2007) credit the failure of many rehabilitation and restoration projects to a failure to consider degrading factors over large enough spatio-temporal scales.

Rehabilitation of entire catchments would be ideal but is rarely practical. However, the condition of the catchment can still inform decisions about which types of rehabilitation to initiate first. As pressures increase in the catchment, the importance of local factors may decline. In catchments experiencing intensive pressure from human development, instream projects should be initiated only after substantial progress in removing major causative factors is made (Schmutz *et al.*, 2007; Booth *et al.*, 2004). Large-scale analyses suggest that rehabilitation prioritization in degraded catchments should be geared toward rehabilitating upslope processes first (e.g., rehabilitating the sediment, wood and water flow regimes; Bohn and Kershner, 2008) or should emphasize rehabilitation of upslope processes in combination with local rehabilitation measures (Schmutz *et al.*, 2007; Wang *et al.*, 2006b; Booth *et al.*, 2004; Soulsby *et al.*, 2001). In relatively undisturbed catchments, local instream habitat and riparian improvement will be most effective (Wang *et al.*, 2006b).

Many kinds of decision tools are now developed to set priorities in catchment-scale rehabilitation planning (e.g., Roni *et al.*, 2008; Steel *et al.*, 2008; Schmutz *et al.*, 2007; Fullerton *et al.*, 2006; Shriver and Randhir, 2006; Reynolds and Peets, 2001). Landscape-scale analyses can inform predictions of future conditions, often through scenario-based tools that integrate landscape evaluations and restoration or rehabilitation planning (e.g., Randhir and Hawes, 2009; Fullerton *et al.*, 2009; Steel *et al.*, 2008; Reynolds and Hessburg, 2005). Decision support frameworks such as the Ecosystem Management Decision Support (EMDS) have been used in salmon recovery planning efforts in the Pacific Northwest USA (Reynolds and Hessburg, 2005). Rieman *et al.* (2001) evaluated the effect of a variety of land management schemes on salmonid viability, and Burnett *et al.* (2007) evaluated how potential habitat available to salmonids is distributed among different land use classes. Using aerial photos, Freeman *et al.* (2003) not only evaluated possible future states in a floodplain, but also evaluated how land use has changed through time using aerial photos. The landscape perspective enables scenario planning to consider far-ranging impacts of particular actions and to provide spatially explicit predictions of trade-offs in future condition that would result from alternative action schemes. The maps of potential future conditions that often result from landscape-based scenarios can be particularly helpful in soliciting citizen input and engaging local landowners (e.g., Baker *et al.*, 2004; Hulse *et al.*, 2004).

Climate change and scenario-based planning

Landscape-scale analyses will be essential in predicting impacts of climate change (e.g., Battin *et al.*, 2007; Rieman *et al.*, 2007; Flebbe *et al.*, 2006) and in identifying management alternatives that are robust to the suite of likely climate change scenarios. Matulla *et al.* (2007) modeled the impact of IPCC (Intergovernmental Panel on Climate Change) emission scenarios on the fish assemblages of the Mur River, Austria. They used downscaled temperature and precipitation predictions to model instream river temperatures over the entire basin in order to identify native species at risk of extirpation and non-native species with increased potential for invasion. Rieman *et al.* (2007) examined potential effects of climate change on bull trout (*Salvelinus confluentus*) over the entire interior Columbia Basin. Working over such large extents, they were able to identify populations of bull trout that faced particularly high potential habitat losses.

In a recently published report of the European Parliament on climate change-induced water stress, land use management was identified as a key issue in adaptation strategies (Anderson *et al.*, 2008). Land use measures that may support adaptation to climate change include, for instance, forestation, conservation agriculture, floodplain rehabilitation, the conversion or rehabilitation of natural land cover, and wetlands rehabilitation. In the Netherlands, projects are being implemented that limit development along rivers to reduce vulnerability to climate change-induced increases in flood risk. The Room for the River Program recognizes the need to widen the river floodplain, rather than increase the height of the dikes (`http://www.ruimtevoorderivier.nl/`).

Monitoring

Large-scale riverine monitoring programs have been driven by the needs of managers and scientists to understand how natural conditions and human impacts vary across watersheds and landscapes (USEPA, 2009; Paulsen *et al.*, 2008; Hughes *et al.*, 2006a; Pont *et al.*, 2006; Yates and Bailey, 2006; Stoddard *et al.*, 2005). Spatially extensive perspectives and landscape analyses have been essential in designing monitoring programs that capture watershed-scale habitat conditions and track performance of entire populations. For example, the USEPA's Environmental Monitoring and Assessment Program (EMAP) began in 1989 and now samples roughly 900 probabilistically selected aquatic sites each year using a rotating panel design (i.e., lakes in 2007, rivers in 2008, streams in 2009, coastal waters in 2010, wetlands in 2011, lakes in 2012, etc.) (Shapiro *et al.*, 2008; Hughes *et al.*, 2000). Biological data, environmental data, and watershed parameters are measured at each site. The probabilistic design and nationally consistent methods allow rigorous statistical inference to all water bodies of each type. Based on the 2005 national wadeable stream survey, Paulsen *et al.* (2008) reported that 42% of all wadeable stream length in the conterminous USA was in poor condition, and the major stressors were nutrients and excess fine sediments. USEPA (2009) reported that 22% of USA lakes were in poor biological condition based on changes in their diatom assemblages, 36% had poor riparian vegetation cover, and 20% were considered to have poor levels of nutrients. A key component in these assessments was the use of large-scale monitoring data.

In 1990, the USGS began its National Water Quality Assessment Program (NAWQA), which focused monitoring and research on 42 study units representing different hydrologic regions and pressures (agriculture, urbanization). The NAWQA program rotates its intensive sampling on 14 of the study units every 3–4 years and proposes to develop models to predict conditions in unmonitored areas (Gilliom *et al.*, 2001). In a 5-year NAWQA study of 9 urban areas, Brown *et al.* (2009) found that the effects of urbanization on fish assemblages differed among those areas. They reported that urbanization affected stream habitats and fish assemblages differently because of natural landscape differences as well as the legacy effects of agriculture.

A major monitoring program (Sustainable Rivers Audit, SRA) for the Murray–Darling Basin in Australia was initiated in 2000, using subbasins as reporting units and systematically rotating annual sampling of 180–341 sites among those basins every 2–3 years (Davies *et al.*, 2006). The SRA stratifies sampling and data analyses by altitudinal zones (lowlands, slopes, uplands, montane) and sites are chosen randomly within each zone to ensure that they are representative.

Large-scale analyses are often applied to monitor the cumulative effectiveness of multiple in-stream rehabilitation (restoration) projects. For several decades a wide variety of agencies within the United States have implemented stream rehabilitation (restoration) projects, but rarely has the effectiveness of those projects been monitored (Alexander and Allan, 2006; Palmer and Bernhardt, 2006; Thompson, 2006; Bernhardt *et al.*, 2005; Roni *et al.*, 2002).

The situation is somewhat better in central Europe where 58% of 50 projects were monitored (Kail *et al.*, 2007). Statistically and ecologically rigorous monitoring of the effectiveness of individual rehabilitation projects is a substantial undertaking and scaling the inference up to evaluate effectiveness at larger scales has rarely been done effectively (Thompson, 2006; Ioannidis, 2005). A key issue in all monitoring programs is the question of sufficient and standardized sampling effort to ensure that site-scale noise or variance among field crews is low relative to the variance or signal from the population of sites being assessed. Standard sampling methods facilitate data compilation and comparison (Bonar *et al.*, 2009; Independent Multidisciplinary Science Team, 2009). Sufficient site extents reduce the local variability that confounds comparison with landscape-scale variables (Flotemersch *et al.*, 2010; Hughes *et al.*, 2008). And high signal-noise ratios increase the potential R-squared values of regression or other statistical analyses when relating assemblage response variables to predictor variables (Stoddard *et al.*, 2008). The 2009 IMST report found that lack

of comparability in survey designs, indicators, and sampling methods hindered data aggregation despite millions of dollars dedicated to disparate research projects in the same area.

5 Opportunities in landscape-scale riverine research

5.1 Moving toward mechanism

Mechanisms linking landform, land use, and climate to instream conditions and fish responses are often hypothesized but we have few formal ways of testing mechanisms that operate over such vast areas. The greatest challenges and, therefore, greatest potential opportunities in the field, clearly lie in our ability to identify and test specific mechanisms. Johnson and Gage (1997) wrote that relationships between the landscape and the stream were not yet well enough defined or quantified as to permit specific predictions of instream responses. This lack of hypotheses has proven a major roadblock to uncovering mechanisms about how streams interact with their surrounding landscapes.

The vast majority of analyses using these data have focused on identifying correlations between landscape conditions and instream conditions, such as fish assemblages, indexes of biological integrity, or water quality parameters. For example, Carlisle *et al.* (2008) correlated biological indices based on fish with urban and agricultural land uses and Pinto *et al.* (2006) found a negative correlation between percent urban area and fish assemblages. Creque *et al.* (2005) linked landscape-scale variables such as mean July temperature with spatial variation in the density of sport fish in Michigan, USA. Often, as Creque *et al.* (2005) did, these analyses compare the relative magnitude of landscape-scale variables versus site-scale variables such as depth, pool distribution, or channel gradient or they compare relationships across similar variables measured at multiple scales (e.g., Feist *et al.*, 2010, 2003; see Table 1 in Durance *et al.*, 2006, for further examples). Many additional examples of correlative analyses linking landscapes and aquatic systems can be found. Those published before 2004 are summarized in Allan (2004a) (see Table 2).

Correlative studies run the risk of assuming causality between things that are merely coincident or that share some common yet unmeasured driver. Allan (2004a) commented that correlative analyses are often plagued by (1) covariation between anthropogenic influences and natural landscape gradients; (2) multiple scale-dependent mechanisms; (3) non-linear relationships; and (4) difficulties in untangling impacts of current versus past conditions. GIS software combined with multi-layer remotely sensed data can generate, literally, hundreds of potential predictor variables for any instream response of interest. As these data are dredged, spurious relationships are bound to arise. Results of the MIRR Project in Austria (Schmutz *et al.*, 2007) show that "uncontrolled observations" of the hydromorphological status of streams have led to numerous uncontrolled and cross-correlated variables. In another example, Dauwalter and Jackson (2004) identified counter intuitive relationships that they explained as the result of random observations over limited ranges of land use and water-quality variables.

One sign of our over-reliance on correlative relationships is our inability to make predictions in new areas. Meta-analyses of studies across regions could elucidate generalities and hone hypotheses. Examining the impacts of agriculture on fish assemblages across disparate basins, ecosystems, and even ecotones would begin to test whether there are, in fact, generalizable relationships between landscapes and instream responses. Brown *et al.* (2009) evaluated the impacts of urbanization on fish assemblages across disparate basins, ecosystems, and ecotones to test for generalizable relationships between landscapes and instream responses, and found few. The development of a standardized GIS framework for collecting, organizing, and sharing riverine and landscape data (Hollenhorst *et al.*, 2007; Brenden *et al.*, 2006) and a standardized framework for effectively incorporating biological data into our digital representations of streams would facilitate such meta-analyses.

Many projects have looked across multiple spatial extents to try and uncover mechanisms linking landscape patterns to instream conditions (e.g., Gido *et al.*, 2006). The underlying concept of this family of analyses has been to compare the strength of correlations at different scales to identify the scale at which the mechanistic relationship exists, and from that, to develop a stronger theory about which mechanisms are causing the observed patterns. Such analyses have been used to identify relationships between land use or geology and fish assemblages in prairie steams in Kansas, USA (Gido *et al.*, 2006); between geology or forest composition and instream habitats such as pool distribution in the forested mountains of the Pacific Northwest, USA (Burnett *et al.*, 2007); and impacts of human disturbance on fish assemblages in flat to rolling topography in the Midwest USA (Wang *et al.*, 2006b). Difficulties of this approach for uncovering mechanisms linking landscape patterns to instream conditions include a reliance on correlative patterns, lack of independence between potential predictor variables within and across scales, and arbitrary assignment of particular variables to a particular scale. For example, Gido *et al.* (2006) assigned stream order to the reach scale whereas some might argue that it could be applied to the catchment or site scale. Despite these limitations, there are many opportunities to improve our mechanistic understanding by incorporating the concept of scale and perhaps expanding it to include both extent and grain and both space and time.

Some progress may be made through application of better and more advanced sampling schemes. Although the technology is not new, advanced statistical sampling designs are increasingly being applied for assessing surface waters in the USA at riverscape (LaVigne *et al.*, 2008a,b), state (Klauda *et al.*, 1998), regional (Ode *et al.*, 2005; Hughes *et al.*, 2004), multi-state (Stoddard *et al.*, 2005; McCormick *et al.*, 2001) and national (USEPA, 2009; Paulsen *et al.*, 2008) scales. Data from these monitoring programs are then used to link biological conditions and trends with landscape stressors. Further consideration of sampling scales for both instream responses and landscape predictors will make such sampling schemes more efficient.

Elucidating causal mechanisms at landscape scales is likely to remain a challenge. Development of more refined hypotheses should limit potential predictors *a priori* and reduce spurious effects. But, relationships between landscape conditions and instream responses are inherently noisy and difficult to model. Not all potential causal factors can be incorporated over large extents. For example, downstream factors are often ignored (Hitt and Angermeier, 2006; Pringle, 1997; Osborne and Wiley, 1992). As well, disturbance thresholds may obscure our ability to detect causal mechanisms (Brendan *et al.*, 2008). Large-scale, long-term experiments would be ideal for uncovering mechanisms (Carpenter *et al.*, 1995); however, these are extremely difficult to manage, usually prohibitively costly and, even when treatments can be applied over entire watersheds, identifying appropriate controls remains a challenge (Strayer *et al.*, 2003). Analysis of existing "natural" experiments (space for time substitution, natural disturbances, before and after policy changes) is a promising approach (e.g., Paulsen *et al.*, 2008; Van Sickle and Paulsen, 2008; Brazner *et al.*, 2007).

5.2 Spatio-temporal structure of human impacts

Human impacts to landscapes occur worldwide (Hughes *et al.*, 2005a; Rinne *et al.*, 2005; Dodge, 1989). Everywhere humans alter landscapes, those landscape are tied to waterbodies that host ecological communities. We are now understanding that human disturbances to river landscapes such as land use (Vitousek *et al.*, 1997, 1986), climate change (Matulla *et al.*, 2007), non-indigenous species (Leprieur *et al.*, 2008b), and morphological and hydrological alterations (Tockner *et al.*, 2009; Nilsson *et al.*, 2005; Tockner and Stanford, 2002; Dynesius and Nilsson, 1994) occur at a global scale. Studies from South America (e.g., Pinto *et al.*, 2006; Tejerina-Garro *et al.*, 2006; Hued and Bistoni, 2003), Asia (e.g., Fausch *et al.*, 2010; An and Choi, 2003; Ganasan and Hughes, 1998; Houssain *et al.*, 2001), Africa (e.g., Kleynhans, 1999; Toham and Teugels, 1999; Hugueny *et al.*, 1996), New Zealand (e.g., Joy and Death, 2004), and Australia (e.g., Davies *et al.*, 2006)

have confirmed that human impacts across the landscape alter fish assemblages.

Many published analyses rely implicitly on the assumption that human activities are randomly distributed across the landscape. But, in fact, human activities are typically constrained by the very same environmental gradients as the biological assemblages of interest. For example, Yates and Bailey (2006) examined relationships between agriculture and landform across 191 basins. They found that agricultural intensity was constrained by geology, in particular drumlin formation and glacial landform type. Covariation between land use and several natural gradients (e.g., geology, soil, slope, elevation, precipitation, temperature) hinders associating land use with biological response (Whittier *et al.*, 2006; Allen *et al.*, 1999). As well, most human disturbances are highly correlated with each other leading to multiple stressors (Fausch *et al.*, 2010) and lack of independence. For example, habitat degradation resulting from hydrological alterations is often accompanied by introductions of non-indigenous fish species that may become invasive, and the effects of these two types of disturbances to river ecosystems are generally impossible to assess separately (Light and Marchetti, 2007; Gurevitch and Padilla, 2004).

Legacy effects of human actions have been well documented in some places (e.g., Harding *et al.*, 1998). Walter and Merritts (2008) detail how current stream form and bed load in many eastern US streams is largely a result of streams down-cutting through pond sediments deposited by many thousands of historic mill dams. And, Poissant *et al.* (2005) found genetic relationships among 12 brook trout populations to better reflect historical hydrologic structure and landscape features than those that were present at the time of the study. Because biological responses can lag behind changes to habitats, poor model fits and spurious results have been linked to a failure to consider historical factors (Van Sickle *et al.*, 2004; Harding *et al.*, 1998). Van Sickle *et al.* (2004) and Harding *et al.* (1998) both reported that historical agricultural land uses seriously limited current assemblage composition. Humphries and Winemiller (2009) describe the limitations of attempting to design riverine restoration projects without incorporating information about the distribution of once abundant fishes. The few existing studies of legacy effects (e.g., Walter and Merritts, 2008) demonstrate that they have a high potential to form constraints for current conditions, but that they are difficult to detect and quantify.

A major opportunity exists in modeling and quantifying the spatio-temporal structure of human impacts at landscape scales. Rather than considering, for example, "agriculture" simply as a predictive variable in isolation, we could consider its spatio-temporal context. We might consider the relationship of the distribution of agriculture to the distribution of urban areas or to the distribution of particular geologies and climatic conditions. We might also consider the relationship of current land use to past agricultural practices and the current distribution and implementation of agriculture to past patterns of human transportation and settlement. Perhaps riverine landscape ecology can benefit from tools developed in the social sciences to better address the complex nature of human impacts to landscapes.

5.3 Statistical opportunities

Johnson and Gage (1997) outlined a series of statistical challenges including (1) skewed data sets, (2) lack of true replication, (3) inherent multi-variate nature of research problems, and (4) colinearity and autocorrelation of landscape metrics. And, Durance *et al.* (2006) noted that large-scale research is often plagued by (1) by non-independent sampling, (2) weak inference, (3) poor model testing or (4) model over-extrapolation. She claimed that these challenges prevent fisheries managers from quantifying the importance of large-scale, anthropogenic disturbances. Pyne *et al.* (2007) and King *et al.* (2005) have also identified serious challenges in relating catchment-scale landscape structure to local-scale biotic processes.

Statistical techniques for testing hypotheses rather than "data mining" have generally been applied despite the lack of strong hypotheses about causal links between landscapes and rivers.

Beale *et al.* (2010) used a simulation study approach to assessing the performance of a suite of statistical models and model fitting methods to synthetic datasets designed to capture the range of issues commonly found in spatial data. They found that the generalized least squares family of models and a Bayesian implementation of the conditional auto-regressive model performed best. While the specific results of their study cannot be applied to all spatial datasets, they provide a good example and a reminder that simulation studies can be used to assess the relative performance of various statistical methods for managing the unique challenges of spatial analysis and can therefore help prevent misuse of techniques such as model selection or *a priori* removal of large-scale spatial trends.

There are also many opportunities for approaching these same data and questions with new or customized statistical tools. Durance *et al.* (2006), for example, suggest that an increased use of geostatistics, popular in landscape ecology, could improve our understanding of the scale-dependence of landscape-fish relationships and improve our ability to test and develop hypotheses about landscape-scale impacts on fish assemblages. Covariance structure analysis (CSA) enabled Wehrly *et al.* (2006) to incorporate both direct and indirect landscape influences in models to predict and understand water temperature patterns in stream networks. Zorn and Wiley (2006) also used CSA to untangle the hierarchy of landscape and local influences on fish biomass distribution in streams. Random forest analysis offers a mechanism for teasing out the major predictor variables from large survey data sets (Cutler *et al.*, 2007; Peters *et al.*, 2007). A more customized technique for linking landscapes and rivers is offered by parametric distance weighting (Van Sickle and Johnson, 2008) which weights landscape and environmental variables by their flowpath distance from the stream. Partitioning variance of the response variable into independent components that reflect spatial variance, environmental variance, noise, and the spatial component of environmental influence is another opportunity (Borcard *et al.*, 1992). Formalized data mining techniques, wavelet analysis, spatial statistics, graph theory (e.g., Schick and Lindley, 2007) and the use of neutral models (e.g., Gardner and Urban, 2007; Gardner *et al.*, 1987) may also help us explore observed patterns and tease out relationships.

To best use results from riverine landscape analyses, we also need to better incorporate and communicate uncertainty into landscape-scale analyses (Burgman *et al.*, 2005). Beginning steps for managing uncertainty include identification of sources of uncertainty, identification (and quantification) of biases that might result from uncertainty, and explicit recognition that management and policy decisions must be robust to known uncertainties. Emerging statistical methods, as described above, may be able to improve the quantification of uncertainty in parameters, in predictions, and across space and time. Mapping methods that can improve our ability to visualize the spatial dimensions of uncertainty will be welcome. The use of alternative scenarios (e.g., Jorgensen *et al.*, 2009; Steel *et al.*, 2008) is also a good tool (Peterson *et al.*, 2003). Alternative scenarios, a method for structured thinking about large-scale patterns in an uncertain and uncontrollable world, are an opportunity both for managing uncertainty and communicating analytical results to inform decision-making.

5.4 Defining the right metrics

Landscapes are inherently heterogeneous and dynamic (Pickett and White, 1985). The idea of spatial heterogeneity, integral to landscape ecology, is not new to aquatic ecology (Thorp *et al.*, 2006; Zalewski *et al.*, 1997). Benda *et al.* (2004) have shown how climatic, hydrologic, and geomorphologic processes mediate the formation and maintenance of dynamic stream habitats. Stream biota, such as salmon (Waples *et al.*, 2008), have evolved in this dynamic landscape. As a result, spatially explicit models have a high potential to address questions about the response of organisms to changing landscape patterns at broad spatial scales (Dunning *et al.*, 1995; Turner *et al.*, 1995). Flitcroft (2007) found hierarchical relationships between dynamic stream habitats and the spatial

distributions of coho salmon. However, adequately capturing these dynamic systems with one or two metrics is a formidable challenge (Uuemaa *et al.*, 2009).

Traditional landscape metrics capture either landscape content (e.g., percent of wetlands within a watershed) or landscape structure (spatial position of land use patches in relation to a waterbody). For instance, does it matter more that urban patches are next to a stream or that there is a large proportion of urban land uses in the watershed relative to forest? Careful application of landscape structure metrics has led to advances in our understanding of many ecological issues (Uuemaa *et al.*, 2009, see Table 1). Examples in riverine ecology include Goetz and Fiske (2008); Van Sickle and Johnson (2008); King *et al.* (2005); Hunsaker and Hughes (2002) and Allan and Johnson (1997). Kearns *et al.* (2005) suggested that the lack of metrics that capture biologically relevant components of spatial pattern is, at least in part, responsible for the mixed conclusions of so many analyses attempting to link land use and instream responses. Therefore, future research should focus on metrics that are able to differentiate landscapes by their configuration as well as heterogeneity and patchiness. Patch density, size, and shape are measures of spatial complexity that could be tested for their ability to better develop linkages between landscape configuration and aquatic responses. Careful development of response metrics is also essential.

There are examples in which the careful development of novel metrics or new indices has advanced our understanding of landscape impacts to rivers. The landscape development index (LDI) of Brown *et al.* (Brown and Vivas, 2005; Brown and Moyle, 2005) uses energy use per unit area to estimate the cumulative impacts of human dominated activities over large extents. By lumping human activities, they have sidestepped the issues in Section 5.2 related to non-independence of human impacts across the landscape. Another example of the use of novel metrics is Scott (2006). He modeled the effects of a trajectory of forest cover change over time on the ratio of endemic specialists to broad-ranged fishes. Using this approach, he quantified the loss of endemism as a result of urbanization.

6 Conclusions

Has new research utilized the strengths of new technologies or are we doing the same old stuff with more expensive data? There are formidable challenges to landscape riverine research but the field is evolving rapidly to meet the needs of both science and management. New technologies have enabled datasets beyond the dreams of most scientists even 30 years ago; but, we remain somewhat hindered by the size and novel nature of these data as well as by our inability to use traditional experimental methods for uncovering meaningful patterns in these data. Spatial data often contain many inter-correlated variables, all of which are potentially of interest, and the uncertainty of these data often has a spatial component that might vary from variable to variable. For example, pool depth might be more accurately measured in larger, wider river systems but hillslope might be more accurately quantified in steeper terrain. As our understanding of the accuracies and inaccuracies of spatial data evolves, we will be better able to utilize use the existing information. We also need to improve our ability to interpret and display spatial data in ways that communicate both accuracy and precision.

Have we incorporated key concepts from landscape ecology to improve our understanding of how landscapes affect rivers? The field of landscape ecology has played a large role in the development of landscape riverine research. The importance of pattern-process interrelationships, connectivity, scale, and fragmentation has guided the development of new work on landscapes and rivers. However, adaptation of these concepts to riverine systems remains a challenge. For example, connectivity in a terrestrial landscape mosaic is quite different from connectivity along a riverine network with unidirectional flow or across an active floodplain with multiple channels. We need to expand our thinking about aquatic connectivity. As another exam-

ple, landscape riverine research will benefit from a broader exploration of the significance of scale to include spatial grain and temporal scale.

Have we been able to use landscape analyses to address management and policy needs? Large-scale analysis of riverine systems has been effective in a wide range of applications. It has been critical in the development of conservation networks and monitoring programs. It has improved our understanding and management of riparian areas in diverse ecosystems. It has led to the improved use of a wide variety of bio-assessment tools. Restoration and rehabilitation prioritization schemes have been enhanced. And lastly, it may be a useful tool for management in the face of climate change. The impact of landscape riverine research has been concentrated in highly developed nations where data and resources are available. As we envisage methods for collecting and combining data across global scales, there will be opportunities for this type of research to affect policies in less developed countries.

Opportunities in landscape-scale riverine research

- There is a need to develop tools for studying mechanistic rather than correlative relationships over large spatial extents. Multi-scale analyses, multi-basin studies, combined process and statistical models, and large-scale experiments can all be harnessed to develop and test generalized theories about how landscape features drive instream responses. Working over larger and larger extents and/or work that leverages long-term data sets may also enable us to increase our sample sizes and generalize results.

- Another opportunity is to shift our perceptions about anthropogenic impacts. First, we must more explicitly model the impacts of economic incentives, human behavior, and population growth patterns on ecological systems. Second, we can benefit from considering the spatio-temporal aspect of human impacts such as the correlation between human development, environmental gradients, and the legacies of past land uses.

- Finally, there are opportunities for a greater number of researchers to incorporate the new statistical tools being designed specifically for linking landscapes to rivers. Quantitative opportunities may also be as simple as defining both landscape and response metrics in novel ways that can more efficiently capture the social, physical, and ecological processes of interest.

7   Acknowledgements

A   Appendix: Landscape perspectives of rivers

Figure I: The Upper Mainstem Willamette River, Oregon, USA. Photo by Steve Cline, USEPA-Corvallis. Despite being channelized in the early 20th century and having multiple hydropower dams in its major tributaries, the upper mainstem of this floodplain river retains channel complexity in the form of side channels, islands, and alcoves.

Figure II: Rio das Velhas, Minas Gerais, Brazil. Photo taken by Carlos Alves, Manuelzao Project. Lacking mainstem or major tributary dams, the das Velhas floods during the rainy season and supports multiple, permanent, off-channel lakes (lagoas) providing reproductive and rearing habitat for aquatic vertebrates.

Figure III: Upper Danube River, Germany. Photo taken by Ashley Steel. This large floodplain river has lost much of its floodplain connectivity as a result of channelization and system-wide dams and locks.

Figure IV: Tagliamento River, Italy. Photo taken by Susanne Muhar. The Tagliamento is one of the largest undammed rivers in western Europe and its braided and anastamosed channels are indicative of high bedload from the Alps.

Figure V: Mekong River and its tributary in Cambodia. Photo taken by Seiichi Nohara during the rainy season. The tributary of this floodplain river is visible only because the riparian forest canopy is barely higher than the water level.

Figure VI: The River Enns, Styria, Austria. Photo from Gesaeuse National Park. Hydropower dams, canals, and channelization limit connectivity of the Enns with its floodplain.

Figure VII: Intermittent tributaries of the Calapooia River, Oregon, USA. Photo by Randy Colvin, Department of Fisheries and Wildlife, Oregon State University. Nearly all of the visible channels (agricultural ditches) are only wet during the rainy season, yet they provide overwintering refuge habitat for spawning and rearing native fish, including salmonids.

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