Aquat. Sci. 64 (2002) 118–128
1015-1621/02/020118-11

Aquatic Sciences

Overview Article

Landscape indicators of human impacts to riverine systems

Sarah E. Gergel[1,]*, Monica G. Turner[2], James R. Miller[2], John M. Melack[3] and Emily H. Stanley[4]

[1] National Center for Ecological Analysis and Synthesis, 735 State St., Suite 300, Santa Barbara, CA 93101, USA
[2] Department of Zoology, 430 Lincoln Dr., University of Wisconsin, Madison, WI 53706, USA
[3] Department of Ecology, Evolution and Marine Biology and Bren School of Environmental Science and Management, University of California, Santa Barbara, CA 93101, USA
[4] Center for Limnology, 600 N. Park St., University of Wisconsin, Madison, WI 53706, USA

Received: 14 November 2001; revised manuscript accepted: 25 February 2002

Abstract. Detecting human impacts on riverine systems is challenging because of the diverse biological, chemical, hydrological and geophysical components that must be assessed. We briefly review the chemical, biotic, hydrologic and physical habitat assessment approaches commonly used in riverine systems. We then discuss how landscape indicators can be used to assess the status of rivers by quantifying land cover changes in the surrounding catchment, and contrast landscape-level indicators with the more traditionally used approaches. Landscape metrics that describe the amount and arrangement of human-altered land in a catchment provide a direct way to measure human impacts and can be correlated with many traditionally used riverine indicators, such as water chemistry and biotic variables. The spatial pattern of riparian habitats may also be an especially powerful landscape indicator because the variation in length, width, and gaps of riparian buffers influences their effectiveness as nutrient sinks. The width of riparian buffers is also related to the diversity of riparian bird species. Landscape indicators incorporating historical land use may also hold promise for predicting and assessing the status of riverine systems. Importantly, the relationship between an aquatic system attribute and a landscape indicator may be nonlinear and thus exhibit threshold responses. This has become especially apparent from landscape indicators quantifying the percent impervious surface (or urban areas) in a watershed, a landscape indicator of hydrologic and geomorphic change.

Key words. Landscape metrics; indicators; human impacts; river.

Introduction

In many countries, legislation mandates assessment of the water chemistry, biota, and physical environment of rivers, many of which have been highly impacted by human activities. For example, the objective of the Clean Water Act of the United States is "to restore and maintain the chemical, physical, and biological integrity of the nations' surface waters." Similarly, the Water Framework Directive of the European Union includes consideration of: (1) biological elements such aquatic flora, benthic invertebrates and fish; (2) hydromorphological elements such as water flow, groundwater dynamics, river depth, width and continuity; and (3) chemical and physiochemical elements such as thermal and oxygenation conditions, salinity, acidification, nutrients, and specific pollutants (Stalzer and Bloch, 2000). Addressing such diverse components poses a serious challenge for monitoring riverine systems.

Landscape ecology emphasizes the interaction between spatial pattern and ecological process (Turner, 1989; Turner et al., 2001) and has conceptual and techni-

* Corresponding author phone: 001 805 892 2500;
fax: 001 805 892 2510; e-mail: gergel@nceas.ucsb.edu
Published on Web: June 19, 2002

cal tools relevant to the monitoring of rivers and their associated catchments. Simple landscape metrics describing the amount of human-altered habitats can be useful indicators of water chemistry, biotic and hydrologic variables. Here, we briefly review traditional riverine assessment approaches and then discuss ways that landscape indicators can be used for assessing the status of river-floodplains. We also compare and contrast landscape-level indicators with traditionally used approaches. We hope to demonstrate how landscape indicators complement traditional riverine indicators, ultimately resulting in an even broader perspective on riverine monitoring.

Traditional approaches to detecting human impacts on rivers

Water chemistry and biotic indices

Chemical indices of water quality are widely used in monitoring (e.g., Dinius, 1987; Smith, 1990; Dojlido et al., 1994; Cude, 2001; Nagels et al., 2001) either as single- or multi-parameter indices. Sets of water chemistry indices have also been developed for community-based monitoring programs. Such indices are typically based on fewer, more conservative water quality attributes such as water clarity, temperature, or conductivity (e.g., Nakamura and Shimatani, 1996; Stewart, 2001). Aquatic invertebrates, algae, plankton and vascular plants have been assessed using integrative field measurements such as the Index of Biotic Integrity (IBI) (Karr, 1981) which combines species richness, composition, trophic structure, abundance, and condition of fish communities into one summary index. Another widely used approach, the River Invertebrate Prediction and Classification System (RIVPACS), was developed primarily for setting conservation priorities. Both RIVPACS and the IBI rely on reference conditions (i.e., a non-impacted or less impacted system) as a benchmark (Karr and Chu, 2000). Biotic indices have been examined further by Karr (1995, 1991) and Rosenberg and Resh (1993).

Instream flow methods

A variety of instream flow methods have been developed to assess hydrologic and hydraulic changes in rivers and generally fall into one of three categories: (a) comparisons between contemporary and historic flows, (b) methods based on hydraulic geometry and (c) instream habitat assessment (Jowett, 1997). Methods relying on historic flows involve analyzing a time series of discharge data, usually obtained from the gauge record or from simulated flows. For example, Richter et al. (1996) developed an Index of Hydrologic Alteration (IHA) to statistically summarize the temporal variability of flows by comparing hydrographs (i.e., plots of discharge versus time) between an unaltered reference system and modified river systems.

Hydraulic instream flow methods relate parameters of hydraulic geometry (such as the wetted perimeter of a river, depth or velocity) to discharge and are widely used to determine minimum allowable flows. The Montana (or Tennant) method is one of the more widely known methods (Tennant, 1976). The critical minimum discharge is often defined by a break or 'inflection point' in discharge curves, below which small decreases in flow result in large decreases in wetted perimeter (Gippel and Stewardson, 1998). Many instream habitat approaches are extensions of hydraulic methods and use wetted perimeter, depth, or velocity to predict areas of suitable habitat for a target species given different flows (Jowett, 1997). Most use univariate habitat suitability curves, and relate variables such as water velocity, minimal water depth, cover, streambed substratum, or water temperature to variation in fish habitat. Because the relationship between flow and habitat suitability is also often non-linear (Jowett, 1997), habitat methods generally identify a threshold flow below which habitat quality or amount declines precipitously (Jowett, 1997).

Physical habitat measures

The International Union of Geological Sciences developed a set of indicators that assess the abiotic environment over broad spatial and temporal scales. Seven of the geoindicators emphasize rivers and include sediment sequence and composition, soil and sediment erosion, stream flow, stream channel morphology, stream sediment storage and load, surface water quality, and the hydrology of floodplains and wetlands. The River Habitat Survey (Fox et al., 1996) developed by the United Kingdom Environment Agency involves surveying the channel, banks, and adjacent land use for river reaches 500 m in length. River data are classified into 9 categories based on geology, gradient and land use and can be used to compare highly modified rivers to those that are 'semi-natural.' Additional physical habitat measures are summarized by Maddock (1999).

The U.S. Environmental Protection Agency has developed Rapid Assessment Protocols similar to the IBI that also incorporates measures of habitat quality such as substratum, in-stream cover, channel morphology, and bank structure. The Index of Stream Condition (Ladson et al., 1999) includes an evaluation of physical features beyond those in the main channel, recognizing the important influence of the surrounding catchment. A review of wetland and riparian approaches is provided by Innis et al. (2000).

Landscape indicators

Many studies have demonstrated that upland land use can influence riverine ecosystems. For example, nutrient losses from many agricultural catchments in the United States are consistently higher than from forested or grassland basins (e.g., Omernik et al., 1981; Johnson et al., 1997). In an upland catchment of the Calado floodplain along the Amazon, Williams et al. (1997) and Williams and Melack (1997) found large increases in solute mobilization from the upper soil horizons after cutting and burning in the catchment. Nutrient ratios in streams were altered from an N to P ratio of 120 : 1 before deforestation to a ratio of 33:1 after deforestation.

Landscape indicators can complement existing approaches for quantifying human impacts on rivers by examining land use change in the surrounding catchment. Landscape indicators quantify the amount and arrangement of land cover, and the physical structure of vegetation on the land surface (Meyer and Turner, 1994). Indicators include the number of cover types present, the proportion of each type on the landscape, the shape of patches and the spatial arrangement and connectivity of patches (Li and Reynolds, 1995). These measures are generally derived from aerial photographs, maps, and satellite imagery using a geographic information system. Land cover is categorized into a set of classes meaningful for a particular study region and scale, often using a hierarchical classification scheme (e.g., Anderson et al., 1976).

Landscape indicators that quantify the amount of and distance to land converted to human uses often explain variability in water chemistry parameters among catchments. For example, the amount of urban land cover and its distance from the stream were the most important variables in predicting N and P concentrations in stream water (Osborne and Wiley, 1988). Lakes with forest-dominated catchments in the Minneapolis-St. Paul area were less eutrophic and had lower levels of chloride and lead than lakes in non-forest dominated catchments (Detenbeck et al., 1993). Many other studies have found relationships between land use and concentrations of nutrients in streams, rivers as well as lakes (Table 1) (e.g., Geier et al., 1994; Hunsaker and Levine, 1995; Johnes et al., 1996; Soranno et al., 1996; Bolstad and Swank, 1997; Johnston et al., 1997; Lowrance, 1998; Bennett et al., 1999).

Landscape indicators that quantify the habitat of riparian zones can also be useful. In many streams, a riparian buffer strip can effectively reduce nitrogen loads to groundwater and phosphorus loads in surface runoff. Furthermore, the spatial pattern of riparian zones may influence their ability to act as nutrient sinks, thus metrics that characterize the spatial pattern of riparian areas can be useful. A simple model of an upland contributing area and a riparian buffer by Weller et al. (1998) explored the relationship between the spatial configuration and nutrient retention of riparian zones. Their heuristic model concluded that spatial characteristics of the buffer were important in determining the buffer's efficacy. Wide riparian zones of uniform width were the most retentive while variable width buffers were less efficient than uniform width buffers because transport through gaps dominated discharge, especially when buffers were narrow. Average buffer width was best predictor of discharge for unretentive buffers and average frequency of gaps was bet predictor for narrow, retentive buffers (Weller et al., 1998). Certainly the efficacy of riparian buffers in any particular setting is influenced by hydrologic flowpaths in addition to the location of a riparian buffer, however.

A riparian buffer versus the entire watershed represent the two extreme scales at which landscape indicators can be measured. Landscape ecology has demonstrated the importance of considering landscape context (that is, characteristics of the surrounding landscape) in addition to local site attributes when explaining local ecological processes. This concept is useful in exploring the variety of scales at which landscape indicators might be most meaningful. For example, when relating landscape indicators to water chemistry, one might calculate the percentage of land surface dedicated to agricultural uses within varying distances (Shuft et al., 1999) or buffers, of stream locations where nitrate was sampled. Statistical analyses would then be used to determine the explanatory power of the landscape metrics at different spatial scales (e.g., Pearson, 1993; Gergel et al., 1999). Nearly all landscape indicators can be measured either at the level of the entire catchment, or in a nearshore riparian zone. This is important as landscape indicators at the scale of the local riparian zone, as well as the scale of the entire catchment, have been shown to be useful predictors water chemistry variables (Table 1).

A variety of investigators have tried to determine the spatial extent, or distance from a water body, over which landscape patterns influence water quality, yet this question remains unresolved (e.g., Omernik et al., 1981; Hunsaker and Levine, 1995; Richards et al., 1996; Johnson et al., 1997; Gergel et al., 1999). McMahon and Harned (1998) found that land use at the scale of an entire catchment was a strong predictor of nutrient concentrations in streams and rivers. In other settings or at different times of the year, the effects of land use on aquatic nutrient loads may be limited more to the immediate vicinity of an aquatic system (e.g., Soranno et al., 1996; Johnson et al., 1997; Cresser et al., 2000).

Variability in aquatic biota can also be explained by landscape indicators measured at the scale of the entire catchment or the scale of the riparian zone (Table 1). For example, in a study of fish in Wisconsin streams, the health of fish communities was negatively correlated

Table 1. Example landscape indicators for monitoring human impacts on various components of riverine ecosystems.

Landscape Indicator	Scale of Measurement	Riverine Component (dependent variable)	Citation
amount of urban land cover distance to stream	catchment	N, P	Osborne and Wiley (1988)
% forest cover % agriculture	catchment	Pb, Cl^-	Detenbeck et al. (1993)
% land use from annual agricultural census	catchment	N, P	Johnes et al. (1996)
% nonforest	catchment	N	Sponseller (2001); Benfield and Valett (2001)
% cover, row crop agriculture and wetlands	catchment	Woody debris	Richards et al. (1996)
forest (positively correlated) agriculture (negatively correlated)	catchment	Fish communities	Wang et al. (1997)
% impervious surface	catchment	Increase in bankfull discharge and surface runoff	Leopold (1968); Arnold and Gibbons (1996)
% impervious surface	catchment	Channel widening	Dunne and Leopold (1978); Booth and Jackson (1997)
% impervious surface	catchment	Fish diversity	Klein (1979); Schueler and Galli (1992)
% impervious surface	catchment	Insect/Invertebrate diversity	Klein (1979)
agriculture	catchment and nearshore scales	$NO_3^- + NO_2^-$, alkalinity, total dissolved solids	Johnson et al. (1997)
% wetlands	catchment and riparian zone	Dissolved organic carbon	Gergel et al. (1999)
% forest	a variety of scales from riparian zone through catchment	Fish and invertebrate diversity	Harding et al. (1998)
average buffer width average frequency of gaps in buffer	riparian	Material discharge (theoretical)	Weller et al. (1998)
% improved riparian zone	riparian	Base cations, alkalinity	Cresser et al. (2000)
land use	nearshore	Nutrients	Johnson et al. (1997)
% nonforest	riparian	Macroinvertebrate density	Sponseller (2001); Benfield and Valett (2001)
% cover, land use	riparian	Sediment	Richards et al. (1996)
length of nonforest along riparian zone	riparian	Fish abundance	Jones et al. (1999)
patch width	riparian	Riparian birds (richness and abundance)	Stauffer and Best (1980); Keller et al. (1993); Darveau et al. (1995); Dickson et al. (1995); Hodges and Krementz (1996)

with the amount of upstream urban development (Wang et al., 1997). The health of fish communities was also positively correlated with amount of upstream forest and negatively correlated with amount of agriculture. This relationship exhibited a nonlinear, threshold response; declines in condition of the fish fauna occurred only after ~20% of the catchment was urbanized, and no impacts were attributed to agriculture until it occupied ~50% of the catchment (Wang et al., 1997). When examining deforestation of riparian zones, Jones et al. (1999) found decreases in fish abundance with increasing length of the nonforested riparian patch. Several species changed dramatically at particular patch lengths (Jones et al., 1999). They suggested that length and area of buffer zones should be emphasized in addition to patch width in mitigation and management (Jones et al., 1999).

The width of the riparian zone can also be used as an indicator for the diversity of riparian bird communities. Woodlands along smaller rivers often form continuous narrow bands, especially in regions where riparian areas are constrained by agriculture or timber harvest (Johnson 1994; Miller et al. 1995). Avian species richness and abundance increases with riparian width (Stauffer and Best, 1980; Keller et al., 1993; Darveau et al., 1995; Dickson et al., 1995; Hodges and Krementz, 1996) and as such, recommendations for the minimum width needed to maintain bird diversity have been made, usually between 50 and 100 m (Croonquist and Brooks, 1993; Keller et al., 1993; Darveau et al., 1995; Hodges and Krementz, 1996).

Quantification of historical land cover can also contribute to useful landscape indicators. Harding et al. (1998) examined fish and invertebrates in 24 catchments in western North Carolina and found that percent forest in the catchment determined from 1950's aerial photographs explained current fish and invertebrate diversity better than land cover from the 1990's. Harding et al. (1998) suggest that in currently forested catchments, historic land use may be a more useful indicator than present land cover.

Finally, the amount of the catchment in impervious surface, or urbanized areas, is a valuable landscape indicator of biotic, hydrologic and geomorphic changes in rivers (see Paul and Meyer, 2001 for a thorough review). Impervious surface cover can influence a variety of hydrologic aspects of streams by shortening the time to flood peaks, causing increases in bankfull discharges and higher surface runoff (Arnold and Gibbons, 1996; Leopold, 1968). Geomorphic changes such as changes in channel width have been associated with percent impervious areas as low as 2–10% (Booth and Jackson, 1997; Morisawa and LaFlure, 1979; Dunne and Leopold, 1978). Initial degradation of fish communities and lower larval densities have been associated with percent impervious areas as low as 10% (Steedman, 1988; Limburg and Schmidt, 1990). It is noteworthy that several thresholds of degradation in streams occur at approximately 10–20% of the catchment in impervious area (Paul and Meyer, 2001).

In summary, landscape patterns influence both biotic and abiotic properties of surface waters and riparian areas. The amount of land converted to human uses (such as agriculture and urban areas) and the arrangement of riparian habitats (patch width or length) are useful indicators of the status of riverine ecosystems. Landscape indicators incorporating historical land use also hold promise for predicting and assessing the status of riverine systems. Table 1 provides even more examples of the utility of landscape indicators for different riverine variables not previously discussed, and the different scales at which landscape indicators are useful.

Comparison of traditional riverine and landscape indicators

Traditional indicators for monitoring streams and rivers have a variety of benefits and weaknesses, as do landscape-level indicators. Next we compare and contrast the advantages and obstacles associated with the different types of indicators (Table 2). We acknowledge that some of our evaluations may be biased towards our experience in North America.

Table 2. Comparison of the general types of indicators used to quantify human impacts on rivers.

Indicator	Advantages	Disadvantages
Chemical Indicators	Direct measure of instream attribute May be quite seasonally variable Delivery may occur at peak flows which may be missed by sampling Citizen monitoring can be economical	Can be hard to collect, store and analyze In community-based sampling, may be prone to error by inexperienced workers
Biotic Indicators	Biotic indicators *may* be able to integrate many changes in watershed conditions over time Indices using fish are relatively easy to identify in the field	Counting invertebrate indicators can be extremely labor intensive and hard to collect Provides qualitative or relative measures May not provide any indication of why a stream is degraded Identification of reference sites can be a challenge, especially for larger rivers
Hydrologic/Hydraulic	Historic flow data is often readily available off the web in the United States Hydraulic habitat methods can relate physical flow to fish, invertebrate habitat Has been expanded to include variables beyond fish habitat	Index of Hydrologic Alteration hasn't been tested in a variety of ecoregional settings Wetted perimeter, for example, has no explicit representation of habitat Considerable field/analytic work is necessary for hydraulic measures
Physical Habitat	Can provide long-term assessment of geomorphic changes Can be assessed at many different scales	Labor intensive due to the variety of spatial scales of interest Measures may not be biologically relevant
Landscape Indicators	Can be linked to other types of indicators Provides a direct measures of human use in a watershed Can assess very large areas Data are often already available across U.S. Data can be stored indefinitely	Requires some training in the use of geographic information systems or aerial photo interpretation Limited to smallest resolution of data The most useful spatial extent of indicators (riparian versus catchment) needs to be established

Chemical monitoring by citizen sampling efforts requires simple, low cost, technically sound procedures for assessing water quality. However, the significance of trends for any single chemical attribute may not be obvious to the general public or policy makers (Cude, 2001; Stewart, 2001). Furthermore, some have argued that water resource management is too focused on water chemistry (Karr, 1995) and that chemical and physical monitoring integrates poorly over time. Biotic indicators may be valuable because they can potentially integrate over many physico-chemical factors acting over extended periods of time (Karr and Chu, 2000), but the relative abilities of chemical versus biotic indicators to integrate watershed conditions over time is not resolved. However, the increased use of biota to assess the status of aquatic ecosystems developed in response to the increasing diversity of stressors and disturbances experienced by these systems, and the inability of traditional chemical remediation to solve a diverse array of problems (Karr, 1991).

Biotic indices (such as the IBI and RIVPACS) have detailed requirements for standardized sampling, laboratory and analytical methods that are essential for consistency. Site scoring often provides only a qualitative ranking, such as excellent, fair, good, or poor, which can be difficult to interpret, particularly when suitable reference rivers are lacking (Harris and Silveira, 1999). Many biotic indices assess the environmental quality of stream reaches without directly considering physical features, and thus may not provide specific information as to why a stream is degraded. Many field-based biotic indicators have focused on, and may be biased towards small streams that are less impacted by humans than larger rivers. For example, RIVPACS-type models have been developed in small, easily wadable streams. Larger lowland rivers are likely more problematic for the development of these types of biotic indicators because large rivers are more degraded (Petts, 1989) and hard to replicate, making the identification of reference sites a challenge.

A challenge in using the Index of Hydrologic Alteration is that the sensitivity and robustness of the parameters used in the index remain to be validated with a variety of human modifications in different ecoregional or physio-geographic settings. Hydraulic instream flow methods that relate parameters of hydraulic geometry (e.g., width, depth, velocity) to discharge are even more challenging to apply than historic flow measures (Jowett, 1997) because of the considerable field and/or analytic work involved. Furthermore, wetted perimeter (or river width), while being a simple, commonly used field-based survey method, has no explicit representation of habitat (Gippel and Stewardson, 1998). None of the instream flow methods address temperature, water quality or biotic interactions, all of which could influence assessments (Jowett, 1997). Many models used for setting instream flow requirements have been criticized as being useful for individual species, but not for understanding the dynamics of a full system with multivariate habitat influences, complex and varied life histories, biotic interactions, and geomorphic changes (Richter et al., 1997). Richter et al. (1997) also suggest that recent advances in understanding natural levels of variability have not been incorporated in instream flow concepts and models.

Many instream habitat methods are extensions of hydraulic methods as they extend the information on wetted perimeter, depth and velocity to predict the areas of suitable habitat for a target species under different flows (Jowett, 1997). Such methods produce an index of suitability of microhabitats, but are rarely related to the actual presence of organisms and have been criticized for merely reducing rivers to a common denominator of habitat for a particular species (Jowett, 1997). However, the historical focus on fish habitat has been recently expanded to include consideration of benthic macroinvertebrates, woody riparian species and recreational goals (Stalnaker et al., 1995). Empirical approaches using regression to analyze relationships between physical features and the biota maybe also be somewhat limited to the particular regions in which they are developed. For example, the Habitat Quality Index (Binns and Eiserman, 1979) relates eleven habitat variables (representing food, shelter, streamflow variation and summer stream temperature) to trout biomass density in Wyoming streams. HABSCORE (Milner et al., 1985) was developed to evaluate salmonid fisheries in Wales. Such regression models tend to be species (and region) specific. As such, the approach, but not the particular models, may be transferable to other areas.

Indicators of water quality, quantity and biological quality are well established compared to indicators using measures such as channel size, channel shape, gradient, bank structure (Maddock, 1999). A particular challenge of assessing physical habitat is the wide range of spatial scales of interest; physical assessment methods must be able to evaluate local microhabitats to habitats at the broader scale of entire river reaches and basins. Broad-scale assessments often involve delineating a reach into shorter segments based on physical characteristics, and sampling involves trade-offs between time and effort and level of detail. The International Union of Geological Sciences (IUGS) geoindicators may be appropriate for detecting local, regional and global change during observational periods of about 100 years (Osterkamp and Schumm, 1996). This could be considered an asset, as it presents a long-term view of river change, or it may be considered less practical for the decision-making time frame of managers and policy-makers.

All the above approaches, when used in some combination, can provide useful and complementary assessments of the state of river-floodplain systems (Table 2).

However, most methods of assessing rivers, whether physical, chemical, or biological, have been developed and used as separate entities. For example, biological assessment has not been well linked with river geomorphology, and this may be partly due to the fact that hydrologists and geomorphologists use different indicators than biologists. Thus, a key consideration for integrating biological indicators with habitat assessment methods is the selection of features that are biologically, and not just geomorphically, relevant (Maddock, 1999). Improved methods would involve concepts from various disciplines to create more holistic indices (Maddock, 1999).

One benefit of landscape-level indicators is that they can be used in conjunction with many traditional instream indicators to expand the consideration of ecological conditions beyond the water's edge, as many of the most-developed indices focus primarily on instream components of riverine systems. Landscape indicators are also useful for larger systems that may be difficult to sample in the field (i. e., non-wadable rivers). The use of Geographic Information Systems has become increasingly common at universities and public agencies and data sets are being created at national, regional, and state levels. Once spatial data are acquired, it can be easily analyzed by technically proficient staff and stored indefinitely. Landscape indicators are useful within the context of regional monitoring schemes that attempt to summarize the status of many aquatic systems over broad geographic regions and entire nations (Jones et al., 2000; Jones et al., 2001). However, the ability of scientists to quantify spatial patterns still exceeds their ability to relate spatial patterns to dynamic ecological processes.

Some real challenges remain evident in the development of landscape indicators of water chemistry. Catchment influences likely differ for different elemental fluxes. For example, in the previous example for Lake Calado along the Amazon, total N yield doubled after land conversion while total P yield increased by a factor of 7. The relative importance of land cover (versus other factors such as geology) may also change seasonally (Johnson et al., 1997), and the predictive power of land use (e. g., R^2) in explaining water chemistry may change seasonally (Gergel et al., 1999). Furthermore, the spatial scale of influence of land use on stream variables may be different for different variables. For example, organic matter inputs may be determined by local conditions (such as vegetative cover at a site) whereas nutrient and sediment delivery may be influenced more by landscape features such as upstream land use (Allan et al., 1997). In another study, water chemistry was related to land cover measured at the scale of the catchment while stream temperature and substratum were related to land cover patterns at the scale of the riparian corridor (or smaller) in the same catchments (Sponseller, 2001; Benfield and Vallett, 2001).

A challenge to ecologists will be to determine the different mechanisms that determine the contributing area within a catchment, i. e., or the scales at which land cover influences aquatic systems across different regions. Variable-source area (VSA) regulation of flushing from soils was proposed by Creed et al. (1996) and Creed and Band (1998 a, b) to explain variations in the export of nitrate from temperate forests. More complex terrain was hypothesized to lead to greater lateral expansion of contributing areas, resulting in longer flushing times and greater nitrate export. Data from catchments in the Sierra Nevada suggest export of nitrate from catchments with greater than 20% soil cover was consistent with the VSA hypothesis, but export from catchments with less than 20% soil cover was not (Sickman, 2001). A better understanding of hydrologic flowpaths and the mechanisms affecting solute delivery to streams may help determine the utility of landscape indicators measured at different spatial scales.

Differences in the natural land cover patterns among different regions may also influence the utility of landscape indicators. For riparian birds, differences in species richness between riverine and upland areas can be less pronounced in semi-arid regions that naturally have a large amount of high-contrast edge (Saab, 1999). For example, in western North America, both interior and edge bird species were found in linear cottonwood (*Populus angustifolia*) patches. The importance of regional differences in land cover is also exemplified by examining nest predation. Higher rates of nest predation have been reported in bottomland hardwoods adjacent to farms (Saracco and Collazo, 1999). However, in the western United States, predation rates were higher in riparian woodlands surrounded by forest than in those surrounded by agricultural fields (Tewksbury et al., 1998). Birds in western U.S. landscapes may be adapted to naturally fragmented habitats, such as linear bands of streamside riparian forests. This is in contrast with regions where previously continuous forest has fragmented by agricultural activity; in such regions, increases in nest predation have repeatedly been correlated with habitat fragmentation (Andrén and Angelstam, 1988; Askins, 1995).

Identifying indicators that provide complementary information about different factors and relate to the condition of catchments (e. g., O'Neill et al., 1997) is difficult, particularly because many indicators of human alteration to river-floodplains focus on instream characteristics. Landscape indicators enable us to consider broader spatial and temporal scales in the development of indicators. Perhaps the best rationale for incorporating landscape indicators into river and stream monitoring is that ecological processes in floodplains affect instream processes (and vice versa), and thus, should not be studied in isolation. Landscape indicators provide the conceptual basis and quantitative tools to link human activities in the catchment to traditional riverine indicators.

Conclusions

Landscape indicators offer new insights about the influences of human activities on aquatic ecosystems that complement those provided by traditional indicators. Landscape indicators characterize attributes of the catchment, particularly relating to the relative proportion and spatial arrangement of natural and human-influenced land cover classes. However, in-stream measurements of limnological variables and aquatic biota remain needed and important. The utility of landscape indicators will depend on the response variable, the strength of the relationship between the indicator and response, and the relative importance of other strong controlling variables (e.g., topographic relief, bedrock, or climate variability). Although the library of empirical studies that relate stream or river quality to landscape metrics is building, additional studies are needed in which both aquatic and landscape variables are quantified simultaneously.

Landscape metrics used as indicators of human influence on river-floodplain ecosystems need not be complex. Rather, simple metrics (e.g., proportion of the catchment or a buffer zone in different land covers and a measure of the arrangement or connectivity of natural and human-modified cover types in the riparian zone) that are straightforward to interpret appear to be effective. The development of landscape metrics was motivated by the premise that ecological processes are related to and can be predicted from aspects of spatial pattern (Baskent and Jordan, 1995; Gustafson, 1998). While research in landscape ecology has produced what often seems a bewildering array of metrics to quantify spatial pattern (e.g., O'Neill et al., 1988; McGarigal and Marks, 1993; Gustafson, 1998; Turner et al., 2001), we suggest that simple metrics may be most useful as complements to other aquatic indicators. Freshwaters are degraded by increasing inputs of silt, nutrients and pollutants from agriculture, forest harvest, and urban areas (Carpenter et al., 1998). Simply determining the amount of natural habitats and these human-altered habitats in a catchment often relates well to concentrations of nutrients and pollutants.

Indicators of the connectivity of riparian habitats may be especially powerful because the spatial pattern of riparian vegetation (i.e., variation in length, width, and gaps) influences its effectiveness as a nutrient sink (Weller et al., 1998). Such measures may be as simple as determining the length (or proportion) of intact riparian vegetation and the number of gaps in that length (e.g., Freeman et al., submitted). However, the most appropriate landscape indicator describing the overall effectiveness of a buffer may be a result of interactions between the spatial arrangement of riparian cover and the localized efficacy of the cover type to act as a nutrient sink. For example, Weller et al. (1998) found that average buffer width was the best predictor of landscape discharge for unretentive buffers, whereas the frequency of gaps was the best predictor for narrow, retentive buffers. The sensitivity of water quality to changes in the riparian zone underscores the need for a spatial view of the catchment.

The relationship between an aquatic system attribute and a landscape indicator may not be linear, and the potential for threshold responses must be considered. Numerous studies in terrestrial systems have suggested non-linear relationships between the proportion of a particular habitat and the persistence and movement of organisms (e.g., Andrén, 1994; With and King 1997) or the spread of disturbance (e.g., Turner et al., 1989; Turner and Romme, 1994). Non-linear relationships have also been found between land cover and fish communities. Similar results have been obtained in other studies that also demonstrated the importance of regional land use as the prime determinant of local stream conditions (e.g., Richards et al., 1996; Allan and Johnson, 1997). The existence of such thresholds has been documented most thoroughly for percent impervious surface in a catchment which is non-linearly related to changes in fish and invertebrate communities, channel widening and surface runoff (Paul and Meyer, 2001). It would be very interesting to know whether similar thresholds are widely applicable for other aquatic variables (Turner et al., in press).

Land-use change is one of the most ubiquitous anthropogenic influences on global ecosystems (Dale et al., 2000). Particularly in North America, land cover patterns have changed dramatically during the past century, and these historic changes may leave persistent legacies. For example, water quality and the structure and function of aquatic ecosystems in the Southern Appalachian region have been strongly influenced by the land-use changes of the 20th century (Harding et al., 1998). While the role of historic land use patterns in determining the current state of aquatic ecosystems is not well studied, this is another area where landscape indicators will be of particular importance. Landscape indicators of past, present and future catchment condition could provide valuable insights into the changing state of aquatic ecosystems.

Studies detecting correlations between river-floodplain condition and landscape-level activities are essential first steps in efficient management of river-floodplain ecosystems. Next steps toward understanding these relationships must include the elucidation of underlying mechanisms governing such relationships (Turner et al., in press). For example, does urbanization negatively influence fishes because it results in too much water or sediment, too little water or sediment, altered nutrients, all of the above, or some other factors? Are there thresholds in the proportion of a particular land use in a catchment beyond which the aquatic system changes qualitatively? Are the spatial scales of landscape influence on aquatic ecosystems similar across systems or site-specific? Understanding when the landscape mosaic is important and identifying the landscape elements critical for particular resources would significantly enhance understanding of river-floodplain ecosystems. Analyses of changes in landscape pattern may prove to be a practical and efficient approach to understanding human impacts in many landscapes (O'Neill et al., 1997).

Acknowledgments

This work was conducted while a Postdoctoral Associate at the National Center for Ecological Analysis and Synthesis, a Center funded by NSF (Grant #DEB-0072909), the University of California, and the Santa Barbara campus. Support for MGT, JRM and EHS was provided by a grant from the U.S. Environmental Protection Agency STAR program (Grant No. R826600). Klement Tockner (and his workshop on Riverine Landscapes in Ascona, Switzerland) also deserve thanks for supporting this work.

References

Allan, J. D., D. L. Erickson and J. Fay, 1997. The influence of catchment land use on stream integrity across multiple spatial scales. Freshwater Biology **37**(1): 149–161.

Allan, J. D. and L. B. Johnson, 1997. Catchment-scale analysis of aquatic ecosystems. Freshwater Biology **37:** 107–111.

Anderson, J. R., E. E. Hardy, J. T. Roach and R. E. Witmer, 1976. A land use and land cover classification system for use with remote sensor data. U.S. Geological Survey Professional Paper 964, 28 pp.

Andrén, H., 1994. Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat. Oikos **71:** 355–366.

Andrén, H. and P. Angelstam, 1988. Elevated predation rates as an edge effect in habitat islands: experimental evidence. Ecology **69:** 544–547.

Arnold, C. L. and C. J. Gibbons, 1996. Impervious surface coverage: the emergence of a key environmental indicator. American Planners Association Journal **62:** 243–58.

Askins, R. A., 1995. Hostile landscapes and the decline of migratory songbirds. Science **267:** 1956–1957.

Baskent, E. Z. and G. A. Jordan, 1995. Characterizing spatial structure of forest landscapes. Canadian Journal of Forest Research **25:** 1830–1849.

Bennett, E. M., T. Reed-Andersen, J. N. Houser, J. R. Gabriel and S. R. Carpenter, 1999. A phosphorus budget for the lake Mendota watershed. Ecosystems **2:** 69–75.

Binns, N. A. and F. M. Eiserman, 1979. Quantification of fluvial trout habitat. Transactions of the American Fisheries Society **108:** 215–228.

Bolstad, P. V. and W. T. Swank, 1997. Cumulative impacts of land use on water quality in a southern Appalachian watershed. Journal of the American Water Resources Association **33:** 519–533.

Booth, D. B. and C. R. Jackson, 1997. Urbanization of aquatic systems: degradation thresholds, stormwater detection, and the limits of mitigation. Journal of the America Water Resources Association **33:** 1077–1090.

Carpenter, S. R., N. F. Caraco, D. L. Correll, R. W. Howarth, A. N. Sharpley and V. H. Smith, 1998. Nonpoint pollution of surface waters with phosphorous and nitrogen. Ecological Applications **8:** 559–568.

Creed, I. F. and L. E. Band, 1998 a. Exploring functional similarity in export of nitrate-N from forested catchments: A mechanistic modeling approach. Water Resources Research **34:** 3079–3093.

Creed, I. F. and L. E. Band, 1998 b. Export of nitrogen from catchments within a temperate forest: Evidence for a unifying mechanism regulated by variable source area dynamics. Water Resources Research **34:** 3105–3120.

Creed, I. F., L. E. Band, N. W. Foster, I. K. Morrison, J. A. Nicolson, R. S. Semkin and D. S. Jeffries, 1996. Regulation of nitrate-N release from temperate forests – a test of the N flushing hypothesis. Water Resources Research **32:** 3337–3354.

Cresser, M. S., R. Smart, M. F. Billett, G. Soulsby, C. Neal, A. Wade, S. Langan and A. C. Edwards, 2000. Modelling water chemistry for a major Scottish river from catchment attributes. Journal of Applied Ecology **37:** 171–184.

Croonquist, M. J. and R. P. Brooks, 1993. Effects of habitat disturbance on bird communities in riparian corridors. Journal of Soil and Water Conservation **48:** 65–70.

Cude, C. G., 2001. Oregon water quality index: a tool for evaluating water quality management effectiveness. Journal of the American Water Resources Association **37:** 125–138.

Dale, V. H., S. Brown, R. Haeuber, N. T. Hobbs, N. Huntly, R. J. Naiman, W. E. Riebsame, M. G. Turner and T. Valone, 2000. Ecological principles and guidelines for managing the use of land. Ecological Applications **10:** 639–670.

Darveau, M., P. Beauchesne, L. Belanger, J. Huot and P. Larue, 1995. Riparian forest strips as habitat for breeding birds in boreal habitat. Journal of Wildlife Management **59:** 67–78.

Detenbeck, N., C. A. Johnston and G. Niemi, 1993. Wetland effects on lake water quality in the Minneapolis/St.Paul metropolitan area. Landscape Ecology **8:** 39–61.

Dickson, J. G., J. H. Williamson, R. N. Conner and B. Ortego, 1995. Streamside zones and breeding birds in east Texas. Wildlife Society Bulletin **23:** 750–755.

Dinius, S. H., 1987. Design of an index of water quality. Water Resources Bulletin **23:** 833–843.

Dojlido, J., J. Raniszewski and J. Woyciechowska, 1994. Water-quality index application for rivers in Vistula River basin in Poland. Water Science and Technology **30:** 57–64.

Dunne, T. and L. B. Leopold, 1978. Water in Environmental Planning. New York, Freeman, 818 pp.

Fox, P. J. A., M. Naura and P. Raven, 1996. Predicting habitat components for semi-natural rivers in the United Kingdom. Proceedings of the 2nd International Conference on Habitats and Hydraulics, Vol B. In: M. Leclerc, H. Capra, S. Valentin, A. Boudreault and I. Cote (eds.), Quebec, CA, pp. 149–162.

Freeman, R. E., E. H. Stanley and M. G. Turner, Submitted. Analysis and conservation implications of landscape change in the floodplain of the Wisconsin River, USA. Ecological Applications.

Geier, T. W., J. A. Perry and L. Queen, 1994. Improving lake riparian source area management using surface and subsurface runoff indices. Environmental Management **18:** 569–586.

Gergel, S. E., M. G. Turner and T. K. Kratz, 1999. Scale-dependent landscape effects on north temperate lakes and rivers. Ecological Applications **9:** 1377–1390.

Gippel, C. J. and M. J. Stewardson, 1998. Use of wetted perimeter in defining minimum environmental flows. Regulated Rivers: Research and Management **14:** 53–67.

Gustafson, E. J., 1998. Quantifying landscape pattern: what is state of the art? Ecosystems **1:** 143–156.

Harding, J. S., E. F. Benfield, P. V. Bolstad, G. S. Helfman and E. B. D. Jones, 1998. Stream biodiversity; the ghost of land use past. Proceedings of the National Academy of Sciences **95:** 14834–14847.

Harris, J. H. and R. Silviera, 1999. Large-scale assessments of river health using an Index of Biotic Integrity with low-diversity fish communities. Freshwater Biology **41:** 235–252.

Hodges, M. F., Jr. and D. G. Krementz, 1996. Neotropical migratory breeding bird communities in riparian forests of different widths along the Altamaha River, Georgia. Wilson Bulletin **108:** 496–506.

Hunsaker, C. T. and D. A. Levine, 1995. Hierarchical approaches to the study of water quality in rivers. Bioscience **45:** 193–203.

Innis, S. A., R. J. Naiman and S. R. Elliot, 2000. Indicators and assessment methods for measuring the ecological integrity of semi-aquatic terrestrial environments. Hydrobiologia **422:** 111–131.

Johnes, P., B. Moss and G. Phillips, 1996. The determination of total nitrogen and total phosphorous concentrations in freshwaters from land use, stock headage and population data: testing a model for use in conservation and watert quality management. Freshwater Biology **36:** 451–473.

Johnson, L. B., C. Richards, G. E. Host and J. W. Arthur, 1997. Landscape influence on water chemistry in Midwestern stream ecosystems. Freshwater Biology **37**(1)**:** 193–218.

Johnson, W. C., 1994. Woodland expansion in the Platte River, Nebraska: patterns and causes. Ecological Monographs **64:** 45–84.

Jones, E. B. D., G. S. Helfman, J. O. Harper and P. V. Bolstad, 1999. Effects of riparian forest removal on fish assemblages in southern Appalachian streams. Conservation Biology **13**(6)**:** 1454–1465.

Jones, K. B., D. T. Heggem, T. G. Wade, A. C. Neale, D. W. Ebert, M. S. Nash, M. H. Mehaffey, K. A. Hermann, A. R. Selle, S. Augustine, I. A. Goodman, J. Pedersen, D. Bolgrien, J. M. Viger, D. Chiang, C. J. Lin, Y. H. Zhong, J. Baker and R. D. van Remortel, 2000. Assessing landscape condition relative to water resources in the western United States: a strategic approach. Environmental Monitoring and Assessment **64**(1)**:** 227–245.

Jones, K. B., A. C. Neale, M. S. Nash, R. D. van Remortel, J. D. Wickham, K. H. Riitters and R. V. O'Neill, 2001. Predicting nutrient and sediment loadings to streams form landscape metrics: a multiple watershed study form the United States Mid-Atlantic Region. Landscape Ecology **16**(4)**:** 301–312.

Jowett, I. G., 1997. Instream flow methods: a comparison of approaches. Regulated Rivers: Research and Management **13:** 115–127.

Karr, J. R., 1981. Assessment of biotic integrity using fish communities. Fisheries **6:** 21–26.

Karr, J. R., 1991. Biological integrity – a long-neglected aspect of water resource management. Ecological Applications **1**(1)**:** 66–84.

Karr, J. R., 1995. Clean water is not enough. Illahee **11:** 51–59.

Karr, J. R. and E. W. Chu, 2000. Sustaining living rivers. Hydrobiologia **422:** 1–14.

Keller, C. M. E., C. S. Robbins and J. S. Hatfield, 1993. Avian communities in riparian forests of different widths in Maryland and Delaware. Wetlands **13:** 137–144.

Klein, R. D., 1979. Urbanization and stream quality impairment. Water Resources Bulletin **15:** 948–963.

Ladson, A. R., L. J. White, J. A. Doolan, B. L. Finlayson, B. T. Hart, P. S. Lake and J. W. Tilleard, 1999. Development and testing of an Index of Stream Condition for waterway management in Australia. Freshwater Biology **41**(2)**:** 453–468.

Leopold, L. B., 1968. Hydrology for Urban Land Planning – A Guidebook on the Effects of Urban Land Use. USGS Circular 554.

Li, H. and J. F. Reynolds, 1995. On definition and quantification of heterogeneity. Oikos **73:** 280–284.

Limburg, K. E. and R. E. Schmidt, 1990. Patterns of fish spawning in Hudson River tributaries: response to an urban gradient? Ecology **71:**1238–45.

Lowrance, R., 1998. Riparian forest ecosystems as filters for nonpoint source pollution. In: M. L. Pace and P. M. Groffman (eds.), *Successes, Limitations and Frontiers in Ecosystem Science*, Springer-Verlag, New York, USA, pp. 113–141.

Maddock, I., 1999. The importance of physical habitat assessment for evaluating river health. Freshwater Biology **41:** 373–391.

McGarigal, K. and B. J. Marks, 1993. FRAGSTATS: Spatial analysis program for quantifying landscape structure. USDA Forest Service General Technical Report PNW-GTR-351.

McMahon, G. and D. A. Harned, 1998. Effect of environmental setting on sediment, nitrogen, and phosphorus concentrations in Albermarle-Pamlico drainage basin, North Carolina and Virginia, USA. Environmental Management **22:** 887–903.

Meyer, W. B. and B. L. Turner, III, (eds.), 1994. *Changes in Land Use and Land Cover: a Global Perspective*. Cambridge University Press, Cambridge, UK. 537 pp.

Miller, J. R., T. T. Schulz, N. T. Hobbs, K. R. Wilson, D. L. Schrupp and W. L. Baker, 1995. Flood regimes and landscape structure: changes in a riparian zone following shifts in stream dynamics. Biological Conservation **72:** 371–379

Milner, N. J., R. J. Hemsworth and B. E. Jones, 1985. Habitat evaluation as a fisheries management tool. Journal of Fish Biology **27**(Suppl A)**:** 85–108.

Morisawa, M. and E. LaFlure, 1979. Hydraulic geometry, stream equilibrium and urbanization. In: D. D. Rhodes and G. P. Williams (eds.), *Adjustments of the Fluvial System*, Dubuque, IA, Kendall-Hunt, pp. 333–50.

Nagels, J. W., R. J. Davies-Colley and D. G. Smith, 2001. A water quality index for contact recreation in New Zealand. Water Science and Technology **43:** 285–292.

Nakamura, K. and Y. Shimatani, 1996. A basic study on comprehensive water indices using glass fiber filters and a spectrophotometer. Water Science and Technology **34:** 163–168.

Omernik, J. M., A. R. Abernathy and L. M. Male, 1981. Stream nutrient levels and proximity of agricultural and forest land to streams: some relationships. Journal of Soil and Water Conservation **36:** 227–231.

O'Neill, R. V., C. T. Hunsaker, K. B. Jones, K. H. Riitters, J. D. Wickham, P. M. Schwartz, I. A. Goodman, B. L. Jackson and W. S. Baillargeon, 1997. Monitoring environmental quality at the landscape scale. BioScience **47:** 513–519.

O'Neill, R. V., J. R. Krummel, R. H. Gardner, G. Sugihara, B. Jackson, D. L. DeAngelis, B. T. Milne, M. G. Turner, B. Zygmunt, S. Christensen, V. H. Dale and R. L. Graham, 1988. Indices of landscape pattern. Landscape Ecology **1:** 153–162.

Osborne, L. L and M. J. Wiley, 1988. Empirical relationships between land use/land cover and stream water quality in an agricultural watershed. Journal of Environmental Management **26:** 9–27.

Osterkamp, W. R. and S. A. Schumm, 1996. Geoindicators for river and river-valley monitoring. *Geoindicators: Assessing Rapid Environmental Changes in Earth Sciences*. In: A. R. Berger and W. J. Iams (eds.) A. A. Balkema, The Netherlands, pp. 97–116.

Paul, M. J. and J. L. Meyer, 2001. Streams in the urban landscape. Annual Review of Ecology and Systematics **32:** 333–365.

Pearson, S. M., 1993. The spatial extent and relative influence of landscape-level factors on wintering bird populations. Landscape Ecology **8:** 3–18.

Petts, G. E., 1989. Historical analysis of fluvial hydrosystems. *Historical Change of Large Alluvial Rivers: Western Europe.* In: G. E. Petts, H. Moller and A. L. Roux (eds.) Wiley, Chichester, pp. 1–18.

Richards, D., L. B. Johnson and G. Host, 1996. Landscape-scale influences on stream habitats and biota. Canadian Journal of Fisheries and Aquatic Science **53**(Suppl. 1): 295–311.

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

Richter, B. D., J. V. Baumgartner and R. Wigington, 1997. How much water does a river need? Freshwater Biology **37:** 231–249.

Rosenberg, D. M. and V. H. Resh, 1993. *Freshwater Biomonitoring and Benthic Macroinvertebrates*, New York, Chapman and Hall.

Saab, V., 1999. Importance of spatial scale to habitat use by breeding birds in riparian forests: a hierarchical analysis. Ecological Applications **9:** 135–151.

Saracco, J. F. and J. A. Collazo, 1999. Predation on artificial nests along three edge types in a North Carolina bottomland hardwood forest. Wilson Bulletin **111:** 541–549.

Schueler, T. R. and J. Galli, 1992. Environmental impacts of stormwater ponds. In: P. Kimble and T. Schueler (eds.), Watershed Restoration Source Book, Washington, DC, Metropol, Wash. Counc. Gov.

Shuft, M. J., T. J. Moser, P. J. Wigington, D. L. Stevens, L. S. McAllister, S. S. Chapman and T. L. Ernst, 1999. Development of landscape metrics for characterizing riparian-stream networks. Photogrammetric Engineering and Remote Sensing **65**(10**):** 1157–1167.

Sickman, J. O., 2001. Comparative analyses of nitrogen biogeochemistry in high-elevation ecosystems. Ph.D. Thesis, University of California, Santa Barbara.

Sponseller, R. A., E. F. Benfield and H. M. Valett, 2001. Relationships between land use, spatial scale and stream macroinvertebrate communities. Freshwater Biology **46**(10**):** 1409–1424.

Smith, D. G., 1990. A better water quality indexing system for rivers and streams. Water Research **24:** 1237–1244.

Soranno, P. A., S. L. Hubler and S. R. Carpenter, 1996. Phosphorous loads to surface waters: a simple model to account for the pattern of land use. Ecological Applications **6**(33**):** 865–878.

Stalnaker, C., B. L. Lamb, J. Henrikson, K. Bovee and J. Bartholow, 1995. The Instream Flow Methodology: A Primer for IFIM. Biological Report 29. US Department of Interior, National Biological Service, Washington DC.

Stalzer, W. and H. Bloch, 2000. Preface: the Austrian Approach. Hydrobiologia **422:** xix-xxi.

Stauffer, D. F. and L. B. Best, 1980. Habitat selection by birds of riparian communities: evaluating effects of habitat alterations. Journal of Wildlife Management **44:** 1–15.

Steedman, R. J., 1988. Modification and assessment of an index of biotic integrity to quantify stream quality in southern Ontario. Canadian Journal of Fisheries and Aquatic Sciences **45:** 492–501.

Stewart, A. J., 2001. A simple stream monitoring technique based on measurements of semiconservative properties of water. Environmental Management **27:** 37–46.

Tennant, D. L., 1976. Instream flow requirements for fish, wildlife and recreation and related environmental resources. In: J. F. Orsborn and C. H. Allman (eds.), Proceedings of the Symposium on Instream Flow Needs II. American Fisheries Society, Bethesda, MD., pp. 359–373.

Tewksbury, J. J., S. Hejl and T. E. Martin, 1998. Breeding productivity does not decline with increasing fragmentation in a western landscape. Ecology **79:** 2890–2903.

Turner, M. G., 1989. Landscape ecology: the effect of pattern on process. Annual Review of Ecology and Systematics **20:** 171–197.

Turner, M. G., R. H. Gardner, V. H. Dale and R. V. O'Neill, 1989. Predicting the spread of disturbance across heterogeneous landscapes. Oikos **55:** 121–129.

Turner, M. G., T. R. Crow, J. Liu, D. Rabe, C. F. Rabeni, P. A. Soranno, W. W. Taylor, K. A. Vogt and J. A. Wiens, In press. Bridging the gap between landscape ecology and natural resource management. In: Liu, J. and W. Taylor (eds.), *Integrating Landscape Ecology into Natural Resource Management.* Cambridge University Press, pp. 433–460.

Turner, M. G., R. H. Gardner and R. V. O'Neill, 2001. *Landscape Ecology in Theory and Practice*. Springer-Verlag, New York.

Turner, M. G. and W. H. Romme, 1994. Landscape dynamics in crown fire ecosystems. Landscape Ecology **9:** 59–77.

Wang, L., J. Lyons, P. Kanehl and R. Gatti, 1997. Influences of watershed land use on habitat quality and biotic integrity in Wisconsin streams. Fisheries **22:** 6–12.

Weller, D. E., T. E. Jordan and D. L. Correll, 1998. Heuristic models for material discharge from landscapes with riparian buffers. Ecological Applications **8:** 1156–1169.

Williams, M. R., T. R. Fisher and J. M. Melack, 1997. Solute dynamics in soil water and groundwater in a central Amazon catchment undergoing deforestation. Biogeochemistry **38:** 303–335.

Williams, M. R. and J. M. Melack, 1997. Solute export from forested and partially deforested catchments in the central Amazon. Biogeochemistry **38:** 67–102.

With, K. A. and A. W. King, 1997. The use and misuse of neutral landscape models in landscape ecology. Oikos **97:** 219–229.

To access this journal online:
http://www.birkhauser.ch