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LANDSCAPES AND RIVERSCAPES: The Influence of Land Use on Stream Ecosystems

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Key Words catchment, disturbance, stressor response, stream health, river

■ **Abstract** Local habitat and biological diversity of streams and rivers are strongly influenced by landform and land use within the surrounding valley at multiple scales. However, empirical associations between land use and stream response only varyingly succeed in implicating pathways of influence. This is the case for a number of reasons, including (*a*) covariation of anthropogenic and natural gradients in the landscape; (*b*) the existence of multiple, scale-dependent mechanisms; (*c*) nonlinear responses; and (*d*) the difficulties of separating present-day from historical influences. Further research is needed that examines responses to land use under different management strategies and that employs response variables that have greater diagnostic value than many of the aggregated measures in current use.

In every respect, the valley rules the stream.
H.B.N. Hynes (1975)

INTRODUCTION

Rivers are increasingly investigated from a landscape perspective, both as landscapes in their own right (Robinson et al. 2002, Ward 1998, Wiens 1989) and as ecosystems that are strongly influenced by their surroundings at multiple scales (Allan et al. 1997, Fausch et al. 2002, Schlosser 1991, Townsend et al. 2003). River ecologists have long recognized that rivers and streams are influenced by the landscapes through which they flow (Hynes 1975, Vannote et al. 1980). However, a landscape perspective of rivers continues to evolve, owing both to the emergence of landscape ecology as a field of study (Turner et al. 2001, Wiens 1989) and to an increased focus on catchment-scale studies by freshwater ecologists.

As Wiens (2002) observed, river ecologists have been doing landscape ecology for a long time but just not calling it that. Landscape ecology places particular emphasis on habitat heterogeneity, connectivity, and scale, all of which have received considerable attention in running waters (Allan 1995). However, most earlier work was conducted at small spatial scales, often within stream reaches of a few hundred meters and their immediate surroundings; less consideration was given to the importance of larger spatial units. Our current understanding of rivers, as with other ecosystems, increasingly incorporates a conceptual framework of spatially nested controlling factors in which climate, geology, and topography at large scales influence the geomorphic processes that shape channels at intermediate scales and thereby create and maintain habitat important to the biota at smaller scales (Allen & Starr 1982, Frissell et al. 1986, Snelder & Biggs 2002). Recognizing that rivers are complex mosaics of habitat types and environmental gradients, characterized by high connectivity and spatial complexity, riverine landscapes increasingly are viewed as "riverscapes" (Fausch et al. 2002, Schlosser 1991, Ward et al. 2002), a unit that is amenable to study over a wide range of scales from a braided river and its valley (Tockner et al. 2002) to small habitat patches (Palmer et al. 2000).

Investigators increasingly recognize that human actions at the landscape scale are a principal threat to the ecological integrity of river ecosystems, impacting habitat, water quality, and the biota via numerous and complex pathways (Allan et al. 1997, Strayer et al. 2003, Townsend et al. 2003). In addition to its direct influences, land use interacts with other anthropogenic drivers that affect the health of stream ecosystems, including climate change (Meyer et al. 1999), invasive species (Scott & Helfman 2001), and dams (Nilsson & Berggren 2000). The recent increase in studies that seek to establish relationships between land use and stream condition is driven by several developments: (*a*) the widespread recognition of the extent and significance of changes in land use and land cover worldwide (Meyer & Turner 1994), (*b*) conceptual and methodological advances in landscape ecology combined with the ready availability of land use/land cover data (Turner et al. 2001), and (*c*) the increasing use of indicators of stream health to assess status and trends of rivers (Karr & Chu 2000, Norris & Thoms 1999).

Hierarchies, Habitats, and Biodiversity

An extensive literature explores the hierarchical nature of river systems, from the largest spatial scale of landscape or basin to successively smaller scales of the valley segment, channel reach, individual channel units (such as riffles and pools), and microhabitat (Figure 1) (Fausch et al. 2002, Frissell et al. 1986, Montgomery 1999). Because stream ecosystems are typically characterized by habitat and biota observed at the scale of a reach, typically $10^1 - 10^3$ m in length, and local species assemblages are strongly influenced by habitat quality and complexity, this geomorphological framework suggests how the stream environment at the local scale is influenced by the surrounding landscape. Reach-level channel morphology is influenced by valley slope and confinement, bed and bank material, and riparian vegetation, as well as by the supply from upslope of water, sediments, and wood (Montgomery & MacDonald 2002). Many features of the dynamic river channel are mutually adjusting (Church 2002), and human activities on the landscape that affect water or sediment supply or that stabilize or destabilize the existing channel shape are likely to set off a complex cascade of changes that are ultimately manifest

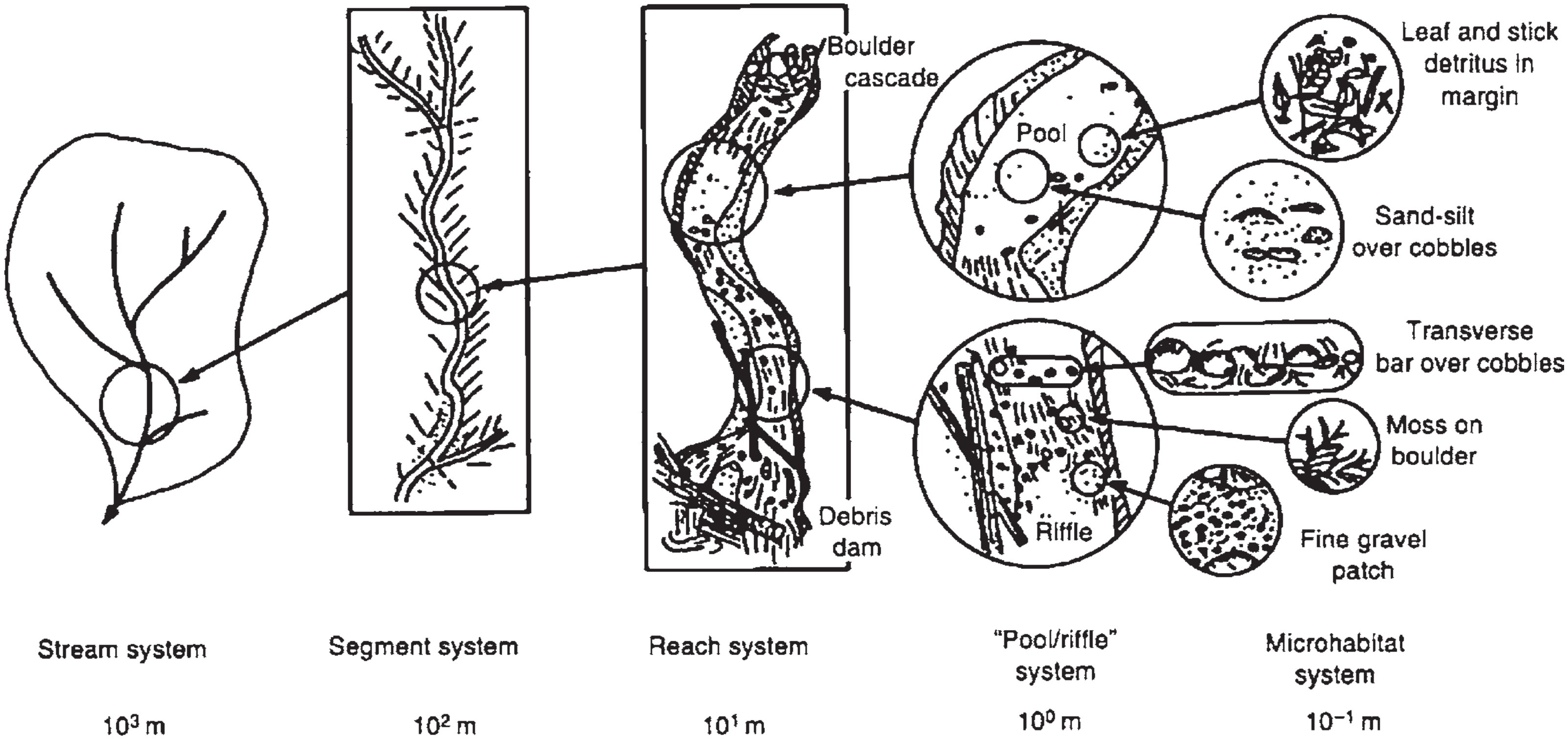

Figure 1 The hierarchical nature of river systems, from the largest spatial scale of landscape or basin to successively smaller scales of the valley segment, channel reach, individual channel units (such as riffles and pools), and microhabitat, as visualized for a small mountain stream (Frissell et al. 1986). Reprinted with permission from *Environmental Management*.

in altered and possibly degraded stream habitat. Natural hydrologic variability, and high flows in particular, move and sort sediments, and through cycles of erosion and deposition create a variety of channel features, including riffles, pools, bars, and islands; cause channel migration; maintain floodplain connectivity and other complex elements of floodplain river channels, including meander loops and side channels; and make the land-water interface both complex and dynamic (Junk et al. 1989). The resultant ever-changing mosaic of habitat patches, ecotones, and successional stages—the riverscape in all its complexity—is largely responsible for the high biodiversity of these systems (Robinson et al. 2002, Ward 1998). In this patch dynamics perspective (Townsend 1989, Hildrew & Giller 1994), the interaction between species-specific habitat needs, life histories, and dispersal ability and the ever-shifting temporal and spatial mosaic of stream habitats support greater diversity than would occur were the habitat unchanging. Thus, both the variety and the variability of habitat are important in influencing the biological diversity of streams and are linked to the larger stream system and surrounding landscape. Human actions at the landscape scale disrupt the geomorphic processes that maintain the riverscape and its associated biota and frequently result in habitat that is both degraded and less heterogeneous.

Assessment of River Health

Ecological integrity, stream condition, and river health are terms that describe the status of fluvial ecosystems and their response to human influences. Condition is defined by similarity of a test site to a set of least-impaired reference sites, whether measured by the sum of several indicators, such as the number of intolerant species and taxa richness [Index of Biotic Integrity (IBI), see Karr 1991], or by the number of observed taxa relative to expected (Rivpacs, see Wright 1995; Ausrivas, see Norris & Hawkins 2000). Additional measures include taxa richness of sensitive species; various biological and ecological traits, such as body size and shape, life history, and behavioral traits (Townsend & Hildrew 1994, Corkum 1999, Usseglio-Polatera et al. 2000b, Richards et al. 1997, Pan et al. 1999); pollution tolerance (Hilsenhoff 1988); and ecosystem processes, such as photosynthesis and respiration (Bunn et al. 1999). Habitat and water quality are also evaluated using individual variables or combined metrics (Barbour et al. 1999). Thus, a wide variety of assessment methods are available to evaluate the response of stream condition to a gradient in land use.

The shape of the relationship between a stream response variable and a measure of stress (Figure 2) likely depends mutually on the sensitivity of the response variable and mode of action of the environmental stressor. A gradual decline indicating incremental change in stream condition might be expected, for example, if a steady increase in sedimentation acted on a species assemblage that was approximately linearly ranked in their sensitivity to sediments. Nonlinear responses are expected whenever the species in question, or the majority of species, exhibit a sensitivity threshold to a particular stress, such as the frequency or magnitude of high flows. Although the response of the biota to some stressors, such as insecticides,

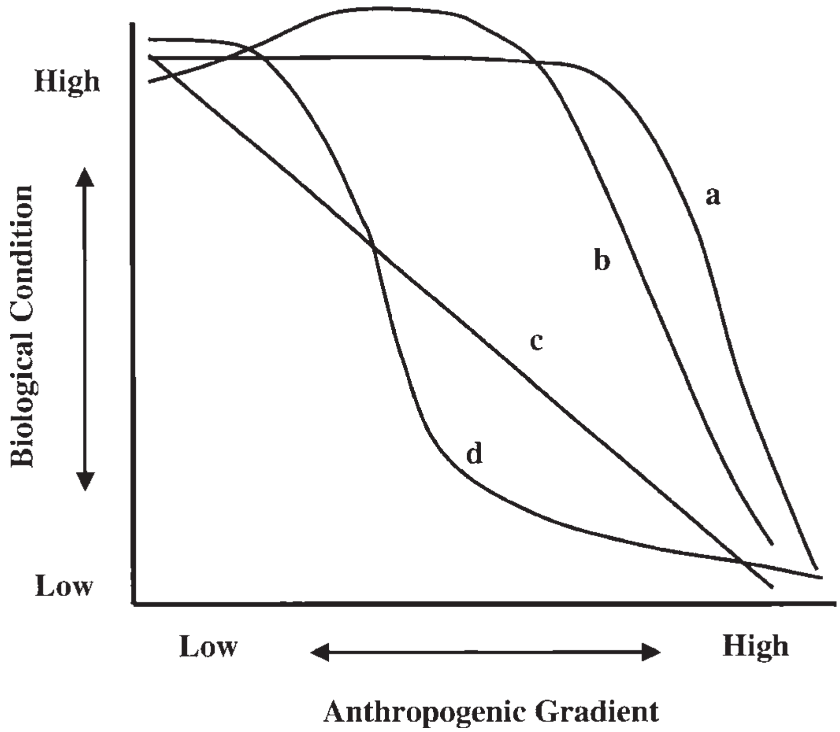

Figure 2 Hypothetical relationship depicting possible responses of stream biological condition (taxon richness, assemblage similarity, or a biological index, scaled to best attainable or reference conditions) to a gradient of increasing environmental stress, measured directly as, e.g., sedimentation, or indirectly as, e.g., agricultural land in the catchment. Possible responses include (*a*) nonlinear response occurring in the high range of the gradient, (*b*) subsidy-stress response, (*c*) linear response, and (*d*) nonlinear (threshold) response occurring in the low range of the gradient. Curves (*a*) versus (*d*) indicate low versus high sensitivity to a stressor. Modified from Norris & Thoms (1999) and Quinn (2000).

is expected to be only negative, a number of environmental stressors have positive influence at low to moderate concentrations. For example, a subsidy-stress response (Odum et al. 1979) may be a common outcome of riparian thinning and a low intensity of agriculture, in which initial increases in light, nutrients, and water temperatures increase periphyton biomass and macroinvertebrate abundance, with no apparent decline in diversity, whereas further intensification of agriculture results in loss of diversity and sensitive species (Quinn 2000).

Ultimately, the range of stream conditions from pristine to profoundly impacted reflects the system's integrated response to various human disturbances acting through the physical space of the catchment hierarchy, over short (pulse) and long (press) durations, and with cascading influences via local habitat structure and food web interactions (Quinn 2000, Townsend & Riley 1999). Different disturbances will exert their influence at different spatial scales and by different pathways (Table 1). Because streams are usually affected by multiple and interacting disturbances, matching a response to the responsible stressor can be very

TABLE 1 Principal mechanisms by which land use influences stream ecosystems

Environmental factor	Effects	References
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	Burkhead & Jelks 2001, Hancock 2002, Henley et al. 2000, Quinn 2000, Sutherland et al. 2002, Walser & Bart 1999, Wood & Armitage 1997
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	Carpenter et al. 1998, Delong & Brusven 1998, Lenat & Crawford 1994, Mainstone & Parr 2002, Niyogi et al. 2003
Contaminant pollution	Increases heavy metals, synthetics, and toxic organics in suspension associated with sediments and in tissues; increases deformities; increases mortality rates and impacts to abundance, drift, and emergence in invertebrates; depresses growth, reproduction, condition, and survival among fishes; disrupts endocrine system; physical avoidance	Clements et al. 2000, Cooper 1993, Kolpin et al. 2002, Liess & Schulz 1999, Rolland 2000, Schulz & Liess 1999, Woodward et al. 1997
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	Allan et al. 1997, Paul & Meyer 2001, Poff & Allan 1995, Walsh et al. 2001, Wang et al. 2001
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	Bourque & Pomeroy 2001, Findlay et al. 2001, Gregory et al. 1991, Gurnell et al. 1995, Lowrance et al. 1984, Martin et al. 1999, Osborne & Kovacic 1993, Stauffer et al. 2000
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	Ehrman & Lamberti 1992, Gurnell et al. 1995, Johnson et al. 2003, Maridet et al. 1995, Stauffer et al. 2000

difficult. Thus, it may be possible to determine the degree of impairment accurately without achieving the same level of certainty regarding cause (Gergel et al. 2002).

THE INFLUENCE OF LAND USE ON RIVERS

The global transition from undisturbed to human-dominated landscapes has impacted ecosystems worldwide and made the quantification of land use/land cover (hereafter, land use) a valuable indicator of the state of ecosystems (Meyer & Turner 1994). Hundreds of studies document statistical associations between land use and measures of stream condition using multisite comparisons and empirical models, and collectively these studies provide strong evidence of the importance of surrounding landscape and human activities to a stream's ecological integrity. Moreover, the extent of land use transformation is staggering. For example, before the development of pastoral agriculture in New Zealand, more than 80% of the land was forested; today, agriculture, primarily the grazing of nearly 60 million sheep and cattle, is the dominant land use in the middle and lower catchment areas of most of New Zealand's streams and rivers (Quinn 2000).

Not surprisingly, agriculture occupies the largest fraction of land area in many developed catchments, whereas urban land use is a much smaller fraction. Of some 150 major river basins of North America, agricultural land use varied from near zero in some northern river systems to 66% in the Upper Mississippi Basin (Benke & Cushing 2004). Six major river basins of the United States have more than 40% of their area in agriculture: the Lower Mississippi, Upper Mississippi, Southern Plains, Ohio, Missouri, and Colorado. Within the Upper Mississippi, the extent of agriculture in large tributary basins varies from 25% in the St. Croix and Wisconsin Rivers to 95% in the Minnesota River Basin. Comparisons of small subcatchments within a larger catchment have reported that the extent of agricultural land use varies even more widely at this smaller spatial scale, from 10% to 70% (Roy et al. 2003), 14% to 99%, (Richards et al. 1996), and 36% to 84% (Roth et al. 1996). Streams draining these landscapes can be expected to experience a wide range of human influences.

Urban land use is commonly a low percentage of total catchment area, yet it exerts a disproportionately large influence both proximately and over distance (Paul & Meyer 2001). Urban land exceeds 5% of catchment area in 29 river basins and exceeds 10% in only 10 of the 150 large basins of North America (Benke & Cushing 2004). However, a large percentage of the land area of small catchments may be urban. Among 30 small (100 km^2) subcatchments of the Etowah Basin, Georgia, combined low- and high-density urban land area averages 15%, with a maximum of 61% (Roy et al. 2003). Impervious surface area reaches as high as 51% for small streams in metropolitan subcatchments of Melbourne, Australia (Walsh et al. 2001), and urban land area is as high as 97% in small catchments of southeastern Wisconsin (Wang et al. 2001).

Other land uses affect stream condition, including forestry, mining, and recreation (Bryce et al. 1999). However, most landscape-scale studies of the influence of land use on streams have contrasted the varying extent of agricultural, urban, and natural (usually forested) land, and so these studies are the primary focus of this review.

Some important caveats apply to studies of the relationship between land use and stream condition. Because land use sums to 100%, several measures of land use may predict stream condition nearly equally well (e.g., Herlihy et al. 1998), and so the interpretation that a particular land use variable is the primary driver of stream condition must be made with caution. Comparisons of land use implicitly substitute space for time, as the often unstated assumption is that locations differing in land use are similar in essentially all other respects and can be viewed as equivalent to the progression over time of a single location experiencing the transition from natural to developed land. Forecasting changes in stream ecosystems in response to changing land use runs the risk that the relationship will change over time owing to changes in specific practices or in the environment itself. For example, revegetation of the riparian to reduce stream temperatures may be negated by future climate change; development of crops with engineered pest resistance may reduce use of pesticides, thus removing one of the pathways by which agricultural land use impacts stream biota.

Agricultural Land Use

Numerous studies have documented declines in water quality, habitat, and biological assemblages as the extent of agricultural land increases within catchments (Richards et al. 1996, Roth et al. 1996, Sponseller et al. 2001, Wang et al. 1997). Researchers commonly report that streams draining agricultural lands support fewer species of sensitive insect and fish taxa than streams draining forested catchments (Genito et al. 2002, Lenat & Crawford 1994, Wang et al. 1997). Although researchers report that row crop and other forms of intensive cultivation strongly affect stream condition, the influence of pasture agriculture may be less pronounced (Meador & Goldstein 2003, Strayer et al. 2003).

Agricultural land use degrades streams by increasing nonpoint inputs of pollutants, impacting riparian and stream channel habitat, and altering flows (Table 1). Higher inputs of sediments, nutrients, and pesticides accompany increased agricultural land use (Cooper 1993, Johnson et al. 1997, Lenat 1984, Osborne & Wiley 1988). Landscape metrics, particularly the proportion of agriculture in the catchment and forest in the riparian zone, explained 65%–84% of the variation in yields of nitrogen, dissolved phosphorus, and suspended sediments for 78 catchments across the five-state Mid-Atlantic Highlands region (Jones et al. 2001). Elevated nutrient concentrations are reported to result in greater algal production and changes in autotroph assemblage composition (Delong & Brusven 1998, Quinn 2000). However, the hypoxic conditions that high nutrient loading causes in lentic and coastal waters (Carpenter et al. 1998) are uncommon in streams and are likely to occur only in localized areas of slow-moving water. Because light levels, nutrient concentrations, and water temperature all tend to increase as riparian forest is lost, algal response may be influenced by one or more of these factors acting in concert. Changes in algal biomass and composition in the upper Roanoke Basin were primarily attributed to light and temperature because nutrients were thought to be sufficient at all sites (Sponseller et al. 2001). Another common response to lost riparian forest is increased macroinvertebrate abundance, particularly grazers, as the food web becomes increasing influenced by autochthonous rather than allochthonous energy sources (Delong & Brusven 1998, Quinn 2000).

Agricultural insecticide and herbicide runoff is likely responsible for some of the association between agricultural land use and stream biota described above (Cooper 1993, Skinner et al. 1997); however, evidence comes primarily from localized toxicity tests rather than from landscape-scale investigations. For example, field enclosures using caged amphipods and laboratory tests that exposed midge larvae to stream sediments showed pesticide toxicity in an agricultural catchment in the United Kingdom (Crane et al. 1996). Furthermore, the disappearance of 8 of the 11 most abundant invertebrate taxa from a reach of headwater stream after surface runoff from arable land was attributed to an insecticide (Schulz & Liess 1999), although most species recovered within 6–11 months, indicating a pulse disturbance. Because the concentrations of agricultural pesticides and herbicides are seldom measured in studies relating agricultural land use to stream biota, their role may be more widespread than is recognized.

Streams in highly agricultural landscapes tend to have poor habitat quality, reflected in declines in habitat indexes and bank stability (Richards et al. 1996, Roth et al. 1996, Wang et al. 1997), as well as greater deposition of sediments on and within the streambed. Sediments in runoff from cultivated land and livestock trampling (Quinn 2000, Strand & Merritt 1999) are considered to be particularly influential in stream impairment (Waters 1995). In the Piedmont region of the Chattahoochee Basin, Georgia, sediments in the channel increased with increasing agricultural land use, while heterogeneity in stream depth and the diversity of fishes associated with coarse substrate in pools declined (Walser & Bart 1999).

Changes to stream hydrology owing to increased agricultural land use are variable, depending on crop evapotranspiration rates compared with natural vegetation, changes to soil infiltration capacity, extent of drainage systems, and, if there is irrigation, whether water is extracted from the river or from groundwater. Mean annual flow of the Kankakee River, Illinois, increased during the twentieth century without any corresponding trend in precipitation, implicating land clearing and urbanization as the cause of greater runoff (Peterson & Kwak 1999). Storm flows commonly increase in magnitude and frequency, especially where runoff is enhanced owing to drainage ditches, subsurface drains, and loss of wetland area. In addition to the impact of flow extremes on erosion and habitat, high flows can eliminate taxa if such events occur during sensitive life stages or with sufficient frequency that only resistant and rapidly dispersing species can tolerate them. Macroinvertebrates that are able to withstand dislodgement or that have short and fast life cycles and good colonizing ability predominated in highly agricultural streams of Michigan (Richards et al. 1997). Alterations to flow regime affect stream fishes by downstream displacement of early life stages and disruption of spawning (Harvey 1987, Schlosser 1985). Although annual and storm flows typically increase with agricultural land use, base flows often decline owing to reduced infiltration and more episodic export of water (Poff et al. 1997). This decline results in an increased area of shallow water habitat, which usually lacks structure and is more easily warmed (Richards et al. 1996).

Wherever agriculture or other anthropogenic activity extends to the stream margin and natural riparian forest is removed, streams are usually warmer during summer and receive fewer energy inputs as leaf litter, and primary production usually increases (Quinn 2000). Bank stability may decrease, although establishment of deep-rooting grasses can stabilize banks (Davies-Colley 1997, Lyons et al. 2000), and the amount of large wood in the stream declines markedly (Johnson et al. 2003). Stable wood substrate in streams performs multiple functions, influencing channel features and local flow and habitat and providing cover for fish, perching habitat for invertebrates, and a substrate for biofilm and algal colonization (Gregory et al. 2003). Its absence can have a profound influence. For example, the presence of wood added an average of 55% and 26% to reach-level local diversity within highly agricultural catchments in Minnesota and Michigan, respectively (Johnson et al. 2003).

Urban Land Use

Substantial changes in biological assemblages are associated with increasing catchment area as urban land (Booth & Jackson 1997, Klauda et al. 1998, Lenat & Crawford 1994, May et al. 1997, Morley & Karr 2002, Tong & Chen 2002, Usseglio-Polatera & Beisel 2002, Wang et al. 2001). Urbanization is the suggested cause of the disappearance of anadromous fishes from tributaries of the Hudson River (Limburg & Schmide 1990). Change in the amount of connected impervious surface was the best single predictor of fish density, diversity, and biotic integrity across a gradient from predominantly agriculture to predominantly urban land in southeastern Wisconsin (Wang et al. 2001). Increasing urbanization among 30 sites within the Etowah Catchment, Georgia, was negatively correlated with water quality, habitat, and measures of the macroinvertebrate assemblage (Roy et al. 2003). Despite the many factors thought to potentially limit Pacific salmon populations, percentage of urban land, along with water quality and sediment flow events, explained more than 60% of the variation in Chinook salmon recruitment in the interior Columbia River Basin from 1980–1990 (Regetz 2003).

Major changes associated with increased urban land area include increases in the amounts and variety of pollutants in runoff, more erratic hydrology owing to increased impervious surface area and runoff conveyance, increased water temperatures owing to loss of riparian vegetation and warming of surface runoff on exposed surfaces, and reduction in channel and habitat structure owing to sediment inputs, bank destabilization, channelization, and restricted interactions between the river and its land margin (Table 1) (Paul & Meyer 2001).

Enhanced runoff from impervious surfaces and stormwater conveyance systems can degrade streams and displace organisms simply because of greater frequency and intensity of floods, erosion of streambeds, and displacement of sediments (Lenat & Crawford 1994). Modeled runoff within the Little Miami Basin, Ohio, was estimated to be more than 55 times greater from impervious than from pervious surfaces (Tong & Chen 2002). A comparison of streams in metropolitan areas and surrounding lands of Melbourne, Australia, found that macroinvertebrate taxa richness declined with increasing impervious surface, but streams of comparable imperviousness were markedly more degraded in the metropolitan drainage system. Flashiness of runoff was considered the primary influence throughout, but the presence of stormwater conveyance systems in the metropolitan area had the added effect of even greater flashiness and the conveyance of multiple pollutants (Walsh et al. 2001).

Whether urban land or impervious surface is a better predictor of the response of stream biota may depend on whether its primary influence is via flow alteration or also involves pollutants. Indeed, biological response measures have been better predicted by impervious area in several landscape studies of stream urbanization (Ourso & Frenzel 2003, Walsh et al. 2001, Wang et al. 2001) and by urban land area in others (Morley & Karr 2002), suggesting that hydrologic influences are primary in some studies, but the broader range of influences represented by urban area may be more important in others.

Because multiple pollutants enter urban streams, the direct influence of particular chemicals and metals is rarely demonstrated in comparisons of urban land use within catchments. Along a steep gradient of urbanization in the vicinity of Anchorage, Alaska, measured as a percentage of impervious area, macroinvertebrate taxa richness declined, and tolerant taxa replaced intolerant taxa (Ourso & Frenzel 2003). Urban land use, chemical factors, channel condition, and instream habitat all correlated with impervious area. However, stream and riparian habitat did not vary as strongly with impervious area as did water and sediment chemistry, suggesting that contaminants may have been of primary importance.

As a cause of changes in the biota, habitat degradation in response to catchment urbanization is less emphasized than other factors, particularly flow variability, although changes to bed sediments are commonly reported (Morley & Karr 2002, Roy et al. 2003). This deemphasis of habitat influence may be because some urban streams have protected corridors that maintain physical habitat but not water quality, or because habitat is relatively uniformly degraded among urbanized sites. Although catchment impervious area was the best single predictor of fish density, diversity, and biotic integrity in southeastern Wisconsin, stream habitat was not well correlated to increasing urbanization, which the authors attributed to prior habitat degradation associated with agriculture (Wang et al. 2001).

FOUR CHALLENGES

Undoubtedly, by changing the landscapes of stream catchments, human activities alter stream ecosystem in multiple ways (Table 1). However, our understanding of the relationships between anthropogenic land use and the ecological integrity of streams is complicated by covariation between anthropogenic and natural gradients, issues of scale, and uncertainties concerning the importance of legacies and thresholds. These challenges are now examined individually, although all may be of importance in a particular catchment study.

Covariation of Anthropogenic and Natural Landscape Features

Gradients of anthropogenic land use are frequently superimposed on an underlying gradient in parent geological material, soil type, topography, and other features of the natural terrain. Anthropogenic and natural factors covary because the latter influences the suitability of locations for agricultural and urban development. Sites near one another tend to be alike in both natural features and human uses, and spatial dependency can be anticipated in the distribution of organisms owing to their habitat requirements and tendency to disperse outwards from locations of high population recruitment (Corkum 1999). Whenever anthropogenic and natural gradients covary and only anthropogenic land use is assessed, the influence attributed to land use can be overestimated.

Whether natural or anthropogenic variables are found to have the stronger effect on stream condition depends substantially on the scope of the study, as well as on the adequate measurement of both types of variables. Nutrient and sediment measures often show that land use overrides natural features, particularly in agricultural lands (Johnson et al. 1997). In Lapwai Creek, an agriculturally impaired stream in northern Idaho, functional groups of macroinvertebrates were similar among sites despite expectations of differences along a river continuum, and the assemblage composition was markedly different from that found in less-impaired streams (Delong & Brusven 1998). Despite substantial variation in terrain and the extent of riparian vegetation, the relative homogeneity of the macroinvertebrate assemblages of these sites was interpreted, via increased sedimentation and the dominance of periphyton as an energy source, as evidence of the overwhelming effect of agricultural land use.

Natural factors may be of primary importance when human influence is minor, or when human influence is widespread and fairly uniform across the study region. In a study of 70 catchments within the relatively undegraded Northern Lakes and Forest Ecoregion of Wisconsin and Michigan, anthropogenic land use was not an important predictor of stream fish assemblages and attributes (Wang et al. 2003). The diversity and abundance of mollusks in the rivers of 36 catchments in Iowa were correlated with landscape factors indicative of erosional and groundwater processes, principally the average percentage of slope and percentage of land area composed of alluvial deposits (Arbuckle & Downing 2002). Possibly because the entire area is highly agricultural, anthropogenic land use was not a predictor. The distribution of mollusks throughout a catchment in southeastern Michigan was well correlated with such local habitat measures as flow stability and substrate, which in turn were more strongly related to geology than to land use (McRae et al. 2004). Although geology and anthropogenic land use both varied throughout the catchment, the former appeared to have the stronger influence over local habitat conditions.

Coequal or at least mixed influence of natural and anthropogenic variables on local stream condition is a frequent finding. Both were influential in explaining patterns in macroinvertebrate assemblages among 55 riffle sites dispersed throughout the Taieri Basin of the South Island of New Zealand (Townsend et al. 2003). Overall, the distribution and abundance of macroinvertebrates were best explained by geomorphological factors at the catchment scale, by a mixture of geomorphology and land use at the reach scale, and predominantly by land use at the bedform (local, riffle) scale. Physical habitat variables explained approximately one third to one half of variation in macroinvertebrate assemblages, depending on season, among 46 sites in the Saginaw Basin of Michigan (Richards et al. 1996). Approximately one half of the habitat variation was explained by landscape variables. Bankfull width and other channel shape measures were much more strongly influenced by geology variables and only minimally by land use. Woody debris showed the opposite partitioning, whereas bankfull depth and canopy cover were equally influenced.

Covariation among natural and anthropogenic environmental factors can make attributing relative influence difficult or impossible. For example, a study of 25 agricultural streams in eastern Wisconsin found that agriculture was a strong influence, but the interpretation was complex owing to covariance of natural and anthropogenic variables measured at multiple spatial scales (Fitzpatrick et al. 2001). Streams with more than 10% agricultural land in their buffers were almost invariably impaired, particularly as indicated by an IBI for fishes, whereas invertebrate and algal metrics were less sensitive to land use. However, because riparian vegetation, geologic conditions, and hydrologic conditions were all correlated with the response of biotic metrics to agricultural land in the catchment, and because the relationships varied with the taxonomic group assessed, researchers could not confidently separate the interrelated effects of geologic setting, catchment and buffer land cover, and base flow.

Spatial Scale

Multiscale investigations often evaluate the relationship between stream condition and land use measured at several of the following scales: (*a*) the local reach, described by a buffer of 100 m to several hundred meters in width on each bank, and some hundreds of meters to a kilometer in length; (*b*) a buffer of similar width but of greater length, often the entire upstream distance for a small stream; and (*c*) the entire catchment upstream of a site (Figure 3). These scales will be referred

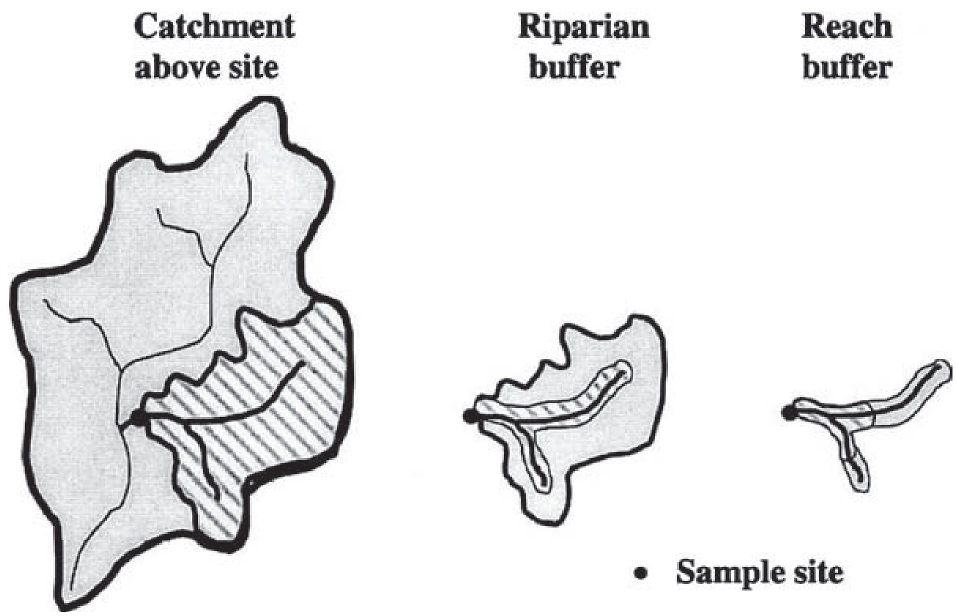

Figure 3 Three spatial scales widely used in relating landscape variables to some physical or biological measure of stream condition. The catchment typically is a subcatchment of a larger basin. Buffer widths of 100–200 m (each bank) are common. Modified from Morely & Karr (2002).

to as reach, riparian, and catchment, respectively; the former clearly is local scale, whereas the latter are two aspects of larger scale.

Different environmental variables of streams can be expected to vary in their responsiveness to large- versus local-scale environmental factors. Shade is influenced by very local patterns of riparian vegetation, water temperature responds to shading over distances of hundreds to thousands of meters, and inputs of leaf litter and wood are local but subject to downstream transport of variable distance (Allan et al. 1997, Quinn 2000). Nutrients and sediments can be transported long distances and so may be influenced by riparian conditions along a stream's entire length. Land use throughout the entire catchment governs stream hydrology through its influence over evapotranspiration, infiltration, and runoff conveyance, and land use is a strong predictor of total nutrient loading (Boyer et al. 2002). The spatial scale at which an effect is detected is influenced by how closely land use in the riparian mirrors land use throughout the catchment, by data resolution, by the interplay of anthropogenic and natural gradients, and by specifics of study design. For example, a comparison of small catchments in southeastern Michigan that spanned a large gradient in agriculture but included only minimal replication of study reaches within catchments found that variation in land use at the catchment scale is the best predictor of stream habitat and fish IBI (Roth et al. 1996). However, reach-scale variation in land use was superior to catchment-scale variation in predicting stream condition within the same river basin when the study design examined multiple reaches within just three small catchments that differed moderately in land use (Lammert & Allan 1999). Variation in land cover is often greater at the reach and riparian scales than at the catchment scale, which likely contributes to the greater influence attributed to riparian land use in many studies (e.g., Stauffer et al. 2000).

When land use at the reach and riparian scales is reported to have a strong influence over stream condition, direct local pathways are usually apparent. Pasture streams with occasional wooded reaches show marked physical and biological changes over distances of less than one kilometer. Forested reaches typically have cooler temperatures, wider channels, fewer sediments, and greater diversity of invertebrates (Abell & Allan 2002, Storey & Cowley 1997, Sweeney 1993). Fish assemblages in Ecuadorian streams changed from dominance by insectivorous and omnivorous taxa in pools with near-stream forest cover to primarily periphyton-grazers in open canopy pools, indicating a direct food web linkage (Bojsen & Barriga 2002). 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).

Even modest riparian deforestation in highly forested catchments can result in degradation of stream habitat owing to sediment inputs. A comparison of two small catchments that were less than 3% nonforested with two that were 13% and 22% nonforested found the latter to have higher concentrations of suspended sediments, higher turbidity at baseflow, five to nine times greater bedload transport, and greater embeddedness (Sutherland et al. 2002). Deforested riparian strips greater than one kilometer in length were associated with more fine sediments and less habitat diversity in a southern Appalachian stream, even though the riparian strips were vegetated and all were located within a highly forested catchment (Jones et al. 1999). Streams with reduced forest cover exhibit declines in overall fish abundance and an increase in sediment-tolerant and invasive species at the expense of those that spawn in clean gravel (Sutherland et al. 2002).

A number of studies have attributed more influence to catchment than to local land use, although pathways of influence may not be as easily detected. A composite index of habitat quality was strongly related to catchment land use and showed progressively weaker associations with riparian and reach-scale land use in southeastern Michigan, where a fish IBI was also more strongly associated with land use throughout the riparian and subcatchment than with reach-scale riparian vegetation (Roth et al. 1996). Invertebrate metrics were better predicted by catchment than by local-scale urbanization in the Puget Sound lowlands of Washington State (Morley & Karr 2002). Catchment-scale influence may be greatest when the primary mechanism is flow instability, nutrients, or some other factor related to the entire landscape.

Studies that examine a variety of measures of stream conditions in relation to land use at multiple scales report, unsurprisingly, mixed influence (Fitzpatrick et al. 2001, Richards et al. 1996, Roth et al. 1996, Stewart et al. 2001). Macroinvertebrate indexes were strongly correlated with both catchment and riparian land cover

over a range of 5%–61% total urban area and 34%–95% forest area in 100-m buffers (Roy et al. 2003). However, macroinvertebrate indexes were even more strongly predicted by environmental factors quantified at the reach-scale, including variation in substrate size and ion concentrations. Because reach-scale conditions were also associated with catchment land cover, these results are consistent with the view that large-scale landscape factors affect the biota via their influence over local-scale physical conditions.

Nonlinearities

Stream condition almost invariably responds nonlinearly to a gradient of increasing urban land or impervious area (IA). A marked decline in species diversity and IBIs with increasing urbanization has been reported from streams in Wisconsin (around 8%–12% IA, Stepenuck et al. 2002, Wang et al. 2000), Delaware (8%–15% IA, Paul & Meyer 2001), Maryland (greater than 12% IA, Klein 1979), and Georgia (15% urban land, Roy et al. 2003). Additional studies (reviewed in Paul & Meyer 2001, Stepenuck et al. 2002) provide evidence of marked changes in discharge, bank and channel erosion, and biotic condition at greater than 10% imperviousness. Although considerable evidence supports a threshold in stream health in the range of 10%–20% IA or urban land, others disagree (Bledsoe & Watson 2001, Karr & Chu 2000), and the relationship is likely too complex for a single threshold to apply. Hydrologic response is influenced by a number of catchment and stream characteristics, including slope, storage, conveyance and connectivity, and channel form (Bledsoe & Watson 2001, Walsh et al. 2001). Also, the supply of contaminants in urban storm runoff may vary independently of impervious area. In contrast to the above studies, a comparison of 45 highly urbanized sites around Seattle, Washington, reported a highly linear decline in macroinvertebrate indexes with increasing urban land and impervious area across the entire gradient (∼10%–60% IA, ∼20%–90% urban land; Morley & Karr 2002).

Streams in agricultural catchments usually remain in good condition until the extent of agriculture is relatively high, more than 30%–50%. In previously forested catchments in New Zealand, a macroinvertebrate fauna typical of undamaged sites was retained and abundances enhanced by conversion of up to 30% of catchment area to pastoral land, but increases in agricultural land above 30% resulted in an increase in pollution-tolerant forms, illustrating a subsidy-stress relationship (Quinn 2000, Quinn & Hickey 1990). In several studies of Wisconsin streams, agricultural land use had a strong effect only when it exceeded 50% of catchment area (Wang et al. 2003, Wang et al. 1997). The response of stream condition to extent of agriculture across 172 sites from 20 major river basins throughout the United States was quite variable, and at least some sites had good fish condition even if agriculture exceeded 50% (Meador & Goldstein 2003). A study of agricultural streams in Wisconsin found indications of a decline in a fish IBI at >30% agriculture in the catchment and >10%–20% agriculture in the buffer (Fitzpatrick et al. 2001); another study reported declines in habitat quality and a fish IBI only when agriculture reached about 50% of catchment area, and some sites maintained high IBI and habitat scores at 80% agriculture (Wang et al. 1997). The wide range of responses reported from streams draining agricultural landscapes clearly indicates that extent of agriculture is not by itself sufficient to predict the strength of the response.

Legacy Effects

Legacy effects are the consequence of disturbances that continue to influence environmental conditions long after the initial appearance of the disturbance. Observing that the present-day diversity of stream macroinvertebrates and fish in forested catchments of the Appalachians, which previously had been farmed, were more similar to streams from present-day agricultural landscapes than from present-day primary forest, Harding et al. (1998) emphasized the importance of the "ghost of land use past." Interpretation of the influence of land use is further complicated when cycles of change occur, such as when agricultural land reverts to forest, or when change is sequential, as when forested land is first converted to agriculture and subsequently to urban land. The finding by Wang et al. (2001) that fish metrics but not habitat varied strongly along an urbanization gradient was interpreted as the legacy of similar habitat degradation at all sites under the common, prior influence of agriculture.

Geomorphological changes brought about by multiple human activities likely have produced lasting, complex, and often unappreciated changes in physical structure and hydrology of river systems. Landscape changes that occurred within a few decades of European settlement of New South Wales, Australia, including clearance of riparian and floodplain vegetation and draining of swamps, have fundamentally altered river structure throughout virtually the entire Bega catchment (Brierley et al. 1999). Extensive habitat transformation has resulted, including channel widening and infilling of pools in lowland sections and incision of headwater channels owing to more efficient downstream water conveyance and downstream export of sediments. Overall structural complexity has been reduced and lateral connectivity is largely lost in middle reaches but is now increased in the lowlands. Unusually, longitudinal connectivity is now greater than was likely true of the presettlement, more discontinuous system. Brierley et al. (1999) estimate that it will take thousands of years for the sediment-starved upper reaches to refill with sediments, while at the same time the oversupply of sand in the lower river is now trapped by exotic vegetation. The timescale of recovery from geomorphic channel alterations is especially long, particularly in comparison to changes in land use, and so stream habitat and channel shape may never reach equilibrium with ongoing development.

How much the channels of some large North American rivers have lost complexity has only recently been appreciated. For example, the Willamette River, Oregon, is estimated to have undergone a fourfold reduction in length of shoreline as its once expansive floodplain and backwaters have been confined to a narrower and simpler channel in response to snag removal, channel dredging, and the draining of its floodplain (Sedell & Froggatt 1984). Under the combined influence of reductions in sediment supply and construction of levees, the Cedar River, Washington, has experienced a 35% decrease in channel width and 45% decrease in channel area (Perkins 1994). Many streams of the upper Midwest have been deprived of wood by past timber harvest and the removal of existing wood in the channel by log drives (Johnson et al. 2003), and headwater regions have been transformed by the removal of beaver and consequent reduction in dams and ponds (Naiman et al. 1988). Elevated concentrations of heavy metals in the water column and in sediments of streams owing to hard-rock mining since the late 1800s were deemed responsible for reduced abundances and diversity of native fishes in northern Idaho (Maret & Maccoy 2002). Legacy effects owing to prior land clearing, channel modifications, snag removal, mining, and perhaps other human actions clearly pose a major challenge to linking present-day land use with concurrent stream condition.

MANAGEMENT APPLICATIONS

The measurement of stream health and its response to a variety of environmental stressors, including land use, requires well-tested indicators of ecological integrity. Composite measures, such as the IBI and percent assemblage similarity, are very useful in detecting overall stream degradation, but because of their aggregated nature they may be less easily interpreted than the behavior of individual response variables (Watzin & McIntosh 1999). For management and restoration actions to be effective, we must diagnose cause as well as assess harm, which requires an improved understanding of the mechanisms through which land use impacts stream ecosystems. Studies are needed that examine the response of individual species, traits, and guilds and that better connect the chain of influence from land use to stream response via studies of mechanisms.

Greater interpretability of stressor-response relationships is usually achieved when the response variables are tolerance groupings, feeding and reproductive guilds, traits, and individual taxa (e.g., Poff & Allan 1995, Townsend et al. 1997, Usseglio-Polatera et al. 2000b). In comparing the responses of several macroinvertebrate metrics with several potential stressors using a large data set from the Mid-Atlantic Highlands, Yuan & Norton (2003) found the proportional abundance of tolerant taxa to be the most sensitive indicator of nutrient enrichment and habitat degradation, whereas Ephemeroptera richness was the most sensitive indicator of high metals and ions. Both linear and nonlinear response relationships were common. Using 11 biological and 11 ecological traits for 472 invertebrate taxa from French rivers, Usseglio et al. (2000a) identified distinct ecological groupings on the basis of body size, reproductive habitat, food source, and feeding habits, an approach that appears to hold promise for bioassessment (Gayraud et al. 2003). Further examples include shifts in fish reproductive guilds in response to sediment inputs (Jones et al. 1999, Sutherland et al. 2002), changes in the relative abundance of species that feed on periphyton versus leaf litter in response to loss of riparian shade (Bojsen & Barriga 2002, Quinn 2000), and an association between taxa with multivoltine life cycles and small body sizes and the extent of shallow, slow-water habitat (Richards et al. 1997). Thus, Poff's (1997) argument that a combination of landscape and habitat filters, together with categorizing or ranking taxa by traits that determine their susceptibility to particular environmental conditions, holds much promise for a multiscale, mechanistic understanding of assemblage response to changing land use and other broad-scale disturbances.

Riparian management is particularly attractive because of the riparian zone's immediate and direct influences on stream condition via well-documented pathways (Gregory et al. 1991, Naiman & Decamps 1997) and because it promises benefits that are highly disproportionate to the land area required (Lowrance et al. 1997, Quinn et al. 2001). However, gaps in the riparian (Weller et al. 1998) as well as subsurface farm and storm drains bypass the riparian zone and diminish its effectiveness (Barton 1996, Osborne & Kovacic 1993). In addition, landscape change at the scale of entire catchments may have impacts too great for a riparian strip to moderate. Studies that evaluate the influence of landscape change across multiple spatial scales report that stream responses are complex and interacting and vary with location and landform setting. For example, nutrient concentrations often reflect catchment land use, whereas macroinvertebrate assemblages appear especially sensitive to a number of local habitat factors (Hunsaker & Levine 1995, Strayer et al. 2003).

Reversal of land use to a less-developed state at the catchment scale is rarely practical, and so improvement of stream condition more often depends on best management practices (BMPs) and improvements in landscape management and design. Some of these activities are at the catchment scale, such as conservation tillage, reduced fertilizer application, and other agricultural BMPs, as well as efforts to minimize hydrologic changes by retaining natural flow paths and infiltration capacity. Other BMPs are more proximate to the stream, such as stormwater retention ponds, managed wetlands, livestock exclusion, and maintenance of an intact riparian corridor. Evaluations of BMP benefits to stream condition commonly report improvements in physical and chemical variables, including habitat, nutrients, sediments, and turbidity (Caruso 2000, D'Arcy & Frost 2001, Lowrance et al. 1997, Strand & Merritt 1999, Wissmar & Beschta 1998). However, studies that evaluate biological responses to BMPs at the scale of the catchment are rare. One such study reported improvements in stream chemistry and streambed sediments in response to buffer practices, but the response of biological metrics was indistinct (Nerbonne & Vondracek 2001; see also Sovell et al. 2000). However, installation of riparian BMPs resulted in improvements in habitat quality and fish abundance in a Wisconsin stream (Wang et al. 2002). More studies of this kind are needed to determine whether physical improvements in stream condition are also evident in the biota.

The ecosystem functions performed by stream riparian zones vary with landform and location, as does human activity within the riparian, and so a "one size fits all" approach to riparian management is unlikely to be effective (Quinn et al. 2001, Strayer et al. 2003). Instead, knowledge of geomorphic setting and the key functions or uses of the riparian that are considered of greatest value should guide riparian management decisions. For example, Lowrance et al. (1997) estimated the amount of sediments, nitrogen, and phosphorus that forested riparian buffers would retain from runoff entering the Chesapeake Bay for each of its nine physiographic provinces. They took into consideration the differences owing to soils, slope, and hydrologic connectivity and predicted different removal efficiencies for different pollutants. Ultimately, BMPs are likely to be chosen on the basis of their demonstrated effectiveness in a particular landform and human setting and of how much society values the expected benefit to the stream ecosystem.

The demonstrated effectiveness of land use data in predicting many components of stream condition points to an expanding role for landscape analysis in catchment management (Gergel et al. 2002). At present, most current studies rely on static Geographic Information Systems (GIS) maps that may represent land cover some years displaced in time from stream condition measures. However, remotely sensed data are likely to become more widely used in the future, offering greater opportunity to synchronize the time frame of land cover and stream condition measurement and to develop new landscape indicators. One promising demonstration showed that stream chemistry, habitat, and stream fish indexes across multiple ecoregions of Nebraska, Kansas, and Missouri were correlated to various "greenness" metrics on the basis of the normalized difference vegetation index, an indicator of vegetation condition and physiological activity obtained from satellite or airborne sensors (Griffith et al. 2002). Although management to mitigate land use impacts on streams will require site-based analysis of interacting factors, detection of areas at risk and estimation of probable risk factors are important and complementary activities to site-based studies.

CONCLUSIONS

The rapidly expanding investigation of streams in the context of their catchments and landscapes clearly indicates that stream ecosystems are strongly affected by human actions across spatial scales. The impacts are numerous, both direct and indirect, and complex, owing to the various pathways by which land use influences streams and the interaction between anthropogenic gradients and the hierarchically structured influence of landform on local stream conditions. Not only does the valley rule the stream, as Hynes (1975) so aptly put it, but increasingly, human activities rule the valley. The extent of change in river health in response to future population growth and development can be anticipated from knowledge of the relationships between land use and stream condition and plausible alternative futures (Baker et al. 2004).

Our understanding of the pathways and mechanisms through which land use influences stream conditions is informed by the comparative and empirical approach that has been the focus of this review; yet, it can also be said that this knowledge at present is extremely limited, particularly for prescriptive management. Our limited understanding is due in part to the multiple effects of a particular change in land use and in part to the influence of local setting and underlying natural variation. Clearly, the influence of the surrounding landscape on a stream is manifest across multiple spatial scales and is further complicated by legacies from prior human activities. Thus, landform apparently operates mainly at the larger scale of catchment and region through its influence over geology, climate, vegetation, and topography, whereas the influence of land use operates across all scales, depending on the response variable of concern. Whether threshold responses are widespread is uncertain, owing partly to the scatter that is common in empirical relationships between land use and stream response. However, impacts of urban land use are clearly experienced at considerably lower percentages of catchment area than is true for agricultural land use, and most studies report a nonlinear response of stream condition to increasing urbanization.

Integrative measures of stream condition, including IBIs and percent similarity measures, are particularly useful for assessing overall stream health because they integrate multiple influences. However, species traits, feeding and reproductive guilds, taxa of known tolerance to particular stressors, and other less-aggregated measures are likely to prove more useful in evaluating pathways and mechanisms (Poff 1997, Usseglio-Polatera et al. 2000a). It will be particularly useful to examine the response of these more sensitive indicators to various management practices intended to offset the harmful impacts of intensive land uses. To date, the majority of catchment-scale studies has only indirectly indicated tradeoffs, as in the common finding that biological metrics are negatively associated with agricultural land in the catchment but positively associated with forested land in the riparian (Steedman 1988, Wang et al. 1997). Future studies that examine the response of more revealing measures such as trait and guild composition, within a two-dimensional matrix of varying land use and management practices, could bring new understanding to the influence of land use on stream condition.

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LITERATURE CITED

Abell RA, Allan JD. 2002. Riparian shade and stream temperatures in an agricultural catchment, Michigan, USA. *Verh. Int. Ver. Theor. Ang. Limnol.* 28:1–6

Allan JD. 1995. *Stream Ecology: Structure and Function of Running Waters*. Dordrecht, Neth.: Kluwer. 388 pp.

Allan JD, Erickson DL, Fay J. 1997. The

influence of catchment land use on stream integrity across multiple spatial scales. *Freshw. Biol.* 37:149–61

Allen TF, Starr TB. 1982. *Hierarchy: Perspectives for Ecological Complexity*. Chicago: Univ. Chicago Press

Arbuckle KE, Downing JA. 2002. Freshwater mussel abundance and species richness: GIS relationships with watershed land use and geology. *Can. J. Fish. Aquat. Sci.* 59:310–16

Baker JP, Hulse DW, Gregory SV, White D, Van Sickle J, et al. 2004. Alternative futures for the Willamette River Basin, Oregon. *Ecol. Appl.* 14:313–24

Barbour MT, Gerritsen J, Snyder BD, Stribling JB. 1999. Rapid Bioassessment Protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates and fish. *Second Edition. Rep. EPA/841-B-99–002*, US EPA, Off. Water, Washington, DC

Barton DR. 1996. The use of Percent Model Affinity to assess the effects of agriculture on benthic invertebrate communities in headwater streams of southern Ontario, Canada. *Freshw. Biol.* 36:397–410

Benke AC, Cushing CE, eds. 2004. *Rivers of North America*. San Diego, CA: Academic/Elsevier

Bledsoe BP, Watson CC. 2001. Effects of urbanization on channel instability. *J. Am. Water Resour. Assoc.* 37:255–70

Bojsen BH, Barriga R. 2002. Effects of deforestation on fish community structure in Ecuadorian Amazon streams. *Freshw. Biol.* 47:2246–60

Booth DB, Jackson CR. 1997. Urbanization of aquatic systems: degradation thresholds, stormwater detection, and the limits of mitigation. *J. Am. Water Resour. Assoc.* 33:311–23

Bourque CPA, Pomeroy JH. 2001. Effects of forest harvesting on summer stream temperatures in New Brunswick, Canada: an inter-catchment, multiple-year comparison. *Hydrol. Earth Syst. Sci.* 5:599–613

Boyer EW, Goodale CL, Jaworski NA, Howarth R. 2002. Anthropogenic nitrogen sources and relationships to riverine nitrogen export in the northeastern U.S.A. *Biogeochemistry* 57/58:137–69

Brierley GJ, Cohen T, Fryirs K, Brooks A. 1999. Post-European changes to the fluvial geomorphology of Bega catchment, Australia: implications for river ecology. *Freshw. Biol.* 41:839–48

Bryce SA, Larsen DP, Hughes RM, Kaufmann P. 1999. Assessing relative risks to aquatic ecosystems: a mid-Appalachian case study. *J. Am. Water Resour. Assoc.* 35:23–36

Bunn SE, Davies PM, Mosisch TD. 1999. Ecosystem measures of river health and their response to riparian and catchment degradation. *Freshw. Biol.* 41:333–46

Burkhead NM, Jelks HL. 2001. Effects of suspended sediment on the reproductive success of the tricolor shiner, a crevice-spawning minnow. *Trans. Am. Fish. Soc.* 130:959–68

Carpenter SR, Caraco NF, Howarth RW, Sharpley AN, Smith VH. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecol. Appl.* 8:559–68

Caruso BS. 2000. Comparative analysis of New Zealand and US approaches for agricultural nonpoint source pollution management. *Environ. Manag.* 25:9–22

Church M. 2002. Geomorphic thresholds in riverine landscapes. *Freshw. Biol.* 47:541–57

Clements WH, Carlisle DM, Lazorchak JM, Johnson PC. 2000. Heavy metals structure benthic communities in Colorado mountain streams. *Ecol. Appl.* 10:626–38

Cooper CM. 1993. Biological effects of agriculturally derived surface water pollutants on aquatic systems—a review. *J. Environ. Q.* 22:402–8

Corkum LD. 1999. Conservation of running waters: beyond riparian vegetation and species richness. *Aquat. Conserv.* 9:559–64

Crane M, Delaney P, Mainstone C, Clarke S. 1996. Measurement by in situ bioassay (on site or on-the-field) of water quality in an agricultural catchment. *Water Res.* 29:2441–48

D'Arcy B, Frost A. 2001. The role of best management practices in alleviating water quality problems associated with diffuse pollution. *Sci. Total Environ.* 265:359–67

Davies-Colley RJ. 1997. Stream channels are narrower in pasture than in forest. *N. Z. J. Mar. Fresh.* 31:599–608

Delong MD, Brusven MA. 1998. Macroinvertebrate community structure along the longitudinal gradient of an agriculturally impacted stream. *Environ. Manag.* 22:445–57

Ehrman TP, Lamberti GA. 1992. Hydraulic and particulate matter retention in a 3rd-order Indiana stream. *J. N. Am. Benthol. Soc.* 11:341–49

Fausch KD, Torgersen CE, Baxter CV, Li HW. 2002. Landscapes to riverscapes: bridging the gap between research and conservation of stream fishes. *BioScience* 52:483–98

Findlay S, Quinn JM, Hickey CW, Burrell G, Downes M. 2001. Effects of land use and riparian flowpath on delivery of dissolved organic carbon to streams. *Limnol. Oceanogr.* 46:345–55

Fitzpatrick FA, Scudder BC, Lenz BN, Sullivan DJ. 2001. Effects of multi-scale environmental characteristics on agricultural stream biota in eastern Wisconsin. *J. Am. Water Resour. Assoc.* 37:1489–507

Frissell CA, Liss WJ, Warren CE, Hurley MD. 1986. A hierarchical framework for stream habitat classification: viewing streams in a watershed context. *Environ. Manag.* 12:199–214

Gayraud S, Statzner B, Bady P, Haybach A, Scholl F, et al. 2003. Invertebrate traits for the biomonitoring of large European rivers: an initial assessment of alternative metrics. *Freshw. Biol.* 48:2045–64

Genito D, Gburek WJ, Sharpley AN. 2002. Response of stream macro invertebrates to agricultural land cover in a small watershed. *J. Freshw. Ecol.* 17:109–19

Gergel SE, Turner MG, Miller JR, Melack JM, Stanley EH. 2002. Landscape indicators of human impacts to riverine systems. *Aquat. Sci.* 64:118–28

Gregory S, Boyer KL, Gurnell AM. 2003. *The ecology and management of wood in world rivers*. Am. Fisheries Soc. Symp. 37, Bethesda, MD

Gregory SV, Swanson FJ, McKee WA, Cummins KW. 1991. An ecosystem perspective of riparian zones: focus on links between land and water. *BioScience* 41:540–51

Griffith JA, Martinko EA, Whistler JL, Price KP. 2002. Interrelationships among landscapes, NDVI, and stream water quality in the US central plains. *Ecol. Appl.* 12:1702–18

Gurnell AM, Gregory KJ, Petts GE. 1995. The role of coarse woody debris in forest aquatic habitats: implications for management. *Aquat. Conserv.* 5:143–66

Hancock PJ. 2002. Human impacts on the stream-groundwater exchange zone. *Environ. Manag.* 29:763–81

Harding JS, Benfield EF, Bolstad PV, Helfman GS, Jones EBD. 1998. Stream biodiversity: the ghost of land use past. *Proc. Natl. Acad. Sci. USA* 95:14843–47

Harvey BC. 1987. Susceptibility of young-of-the-year fishes to downstream displacement by flooding. *Trans. Am. Fish. Soc.* 116:851–55

Henley WF, Patterson MA, Neves RJ, Lemly AD. 2000. Effects of sedimentation and turbidity on lotic food webs: a concise review for natural resource managers. *Rev. Fish. Sci.* 8:125–39

Herlihy A, Stoddard JL, Johnson CB. 1998. The relationship between stream chemistry and watershed land cover data in the Mid-Atlantic region, USA. *Water Air Soil Pollut.* 105:377–86

Hildrew AG, Giller PS. 1994. Patchiness, species interactions and disturbance in the stream benthos. In *Aquatic Ecology: Scale, Pattern and Process*, ed. PS Giller, AG Hildrew, DG Raffaelli, pp. 21–62. Oxford, UK: Blackwell

Hilsenhoff WL. 1988. Rapid field assessment of organic pollution with a family-level biotic index. *J. N. Am. Benthol. Soc.* 7:65–68

Hunsaker CT, Levine DA. 1995. Hierarchical approaches to the study of water-quality in rivers. *BioScience* 45:193–203

Hynes HBN. 1975. The stream and its valley. *Verh. Int. Ver. Theor. Ang. Limnol.* 19:1–15

Johnson LB, Breneman DH, Richards C. 2003. Macroinvertebrate community structure and function associated with large wood in low gradient streams. *River Res. Appl.* 19:199–218

Johnson LB, Richards C, Host GE, Arthur JW. 1997. Landscape influences on water chemistry in Midwestern stream ecosystems. *Freshw. Biol.* 37:193–208

Jones EBD, Helfman GS, Harper JO, Bolstad PV. 1999. Effects of riparian forest removal on fish assemblages in southern Appalachian streams. *J. Environ. Manag.* 13:1454–65

Jones KB, Neale AC, Nash MS, Van Remortel RD, Wickham JD, et al. 2001. Predicting nutrient and sediment loadings to streams from landscape metrics: a multiple watershed study from the United States Mid-Atlantic Region. *Landsc. Ecol.* 16:301–12

Junk WJ, Bayley PB, Sparks RE. 1989. The flood pulse concept in river-floodplain systems. In *Proc. Int. Large River Symp., Can. Spec. Publ. Fish. Aquat. Sci.*, ed. DP Dodge, 106:110–27

Karr JR. 1991. Biological integrity—a long-neglected aspect of water-resource management. *Ecol. Appl.* 1:66–84

Karr JR, Chu EW. 2000. Sustaining living rivers. *Hydrobiologia* 422/423:1–14

Klauda R, Kazyak P, Stranko S, Southerland MT, Roth NE, Chaillou J. 1998. Maryland biological stream survey: a state agency program to assess the impact on anthropogenic stress on stream habitat quality and biota. *Environ. Monit. Assess.* 51:299–316

Klein RD. 1979. Urbanization and stream water quality impairment. *Water Resour. Bull.* 15:948–63

Kolpin DW, Furlong ET, Meyer MT, Thurman EM, Zaugg SD, et al. 2002. Pharmaceuticals, hormones, and other organic wastewater contaminants in US streams, 1999–2000: a national reconnaissance. *Environ. Sci. Technol.* 36:1202–11

Lammert M, Allan JD. 1999. Assessing biotic integrity of streams: effects of scale in measuring the influence of land use/cover and habitat structure on fish and macroinvertebrates. *Environ. Manag.* 23:257–70

Lenat DR. 1984. Agriculture and stream water-quality—a biological evaluation of erosion control practices. *Environ. Manag.* 8:333–43

Lenat DR, Crawford JK. 1994. Effects of land-use on water-quality and aquatic biota of three North Carolina Piedmont streams. *Hydrobiologia* 294:185–99

Liess M, Schulz R. 1999. Linking insecticide contamination and population response in an agricultural stream. *Environ. Toxicol. Chem.* 18:1948–55

Limburg KE, Schmide RE. 1990. Patterns of fish spawning in the Hudson River tributaries: Response to an urban gradient? *Ecology* 71:1238–45

Lowrance R, Altier LS, Newbold JD, Schnabel RR, Groffman PM, et al. 1997. Water quality functions of riparian forest buffers in Chesapeake Bay watersheds. *Environ. Manag.* 21:687–712

Lowrance R, Todd R, Fail JJ, Hendrickson OJ, Leonard R, Asmussen L. 1984. Riparian forests as nutrient filters in agricultural watersheds. *BioScience* 34:374–77

Lyons J, Trimble SW, Paine LK. 2000. Grass versus trees: managing riparian areas to benefit streams of central North America. *J. Am. Water Resour. Assoc.* 36:919–30

Mainstone CP, Parr W. 2002. Phosphorus in rivers—ecology and management. *Sci. Total Environ.* 282:25–47

Maret TR, Maccoy DE. 2002. Fish assemblages and environmental variables associated with hard-rock mining in the Coeur d'Alene River Basin, Idaho. *Trans. Am. Fish. Soc.* 131:865–84

Maridet L, Wasson JG, Philippe M, Amoros C. 1995. Benthic organic-matter dynamics in three streams—riparian vegetation or bed morphology control. *Arch. Hydrobiol.* 132:415–25

Martin TL, Kaushik NK, Trevors JT, Whiteley HR. 1999. Review: denitrification in temperate climate riparian zones. *Water Air Soil Poll.* 111:171–86

May CW, Horner RR, Karr JR, Mat BW, Welch EB. 1997. Effects of urbanization on small streams in the Puget Sound Lowland Ecoregion. *Watershed Prot. Tech.* 2:485–94

McRae SE, Allan DJ, Burch JB. 2004. Reach- and catchment-scale determinants of distribution of freshwater mussels (Bivalvia: Unionidae) in south-eastern Michigan, USA. *Freshw. Biol.* 49:127–42

Meador MR, Goldstein RM. 2003. Assessing water quality at large geographic scales: relations among land use, water physicochemistry, riparian condition, and fish community structure. *Environ. Manag.* 31:504–17

Meyer JL, Sale MJ, Mulholland PJ, Poff NL. 1999. Impacts of climate change on aquatic ecosystem functioning and health. *J. Am. Water Resour. Assoc.* 35:1373–86

Meyer WB, Turner BL, eds. 1994. *Changes in Land Use and Land Cover: A Global Perspective*. New York: Cambridge Univ. Press. 537 pp.

Montgomery DR. 1999. Process domains and the river continuum. *J. Am. Water Resour. Assoc.* 35:397–410

Montgomery DR, MacDonald LH. 2002. Diagnostic approach to stream channel assessment and monitoring. *J. Am. Water Resour. Assoc.* 38:1–16

Morley SA, Karr JR. 2002. Assessing and restoring the health of urban streams in the Puget Sound Basin. *Conserv. Biol.* 16:1498–509

Naiman RJ, Decamps H. 1997. The ecology of interfaces: riparian zones. *Annu. Rev. Ecol. Syst.* 28:621–58

Naiman RJ, Johnston CA, Kelley JC. 1988. Alteration of North American streams by beaver. *BioScience* 38:753–62

Nerbonne BA, Vondracek B. 2001. Effects of local land use on physical habitat, benthic macroinvertebrates, and fish in the Whitewater River, Minnesota, USA. *Environ. Manag.* 28:87–99

Nilsson C, Berggren K. 2000. Alterations of riparian ecosystems caused by river regulation. *BioScience* 50:783–92

Niyogi DK, Simon KS, Townsend CR. 2003. Breakdown of tussock grass in streams along a gradient of agricultural development in New Zealand. *Freshw. Biol.* 48:1698–708

Norris RH, Hawkins CP. 2000. Monitoring river health. *Hydrobiologia* 435:5–17

Norris RH, Thoms MC. 1999. What is river health? *Freshw. Biol.* 41:197–209

Odum EP, Finn JT, Franz EH. 1979. Perturbation theory and the subsidy-stress gradient. *BioScience* 29:349–52

Osborne LL, Kovacic DA. 1993. Riparian vegetated buffer strips in water-quality restoration and stream management. *Freshw. Biol.* 29:243–58

Osborne LL, Wiley MJ. 1988. Empirical relationships between land use/cover and stream water quality in an agricultural watershed. *J. Environ. Manag.* 26:9–27

Ourso RT, Frenzel SA. 2003. Identification of linear and threshold responses in streams along a gradient of urbanization in Anchorage, Alaska. *Hydrobiologia* 501:117–31

Palmer MA, Swan CM, Nelson K, Silver P, Alvestad R. 2000. Streambed landscapes: evidence that stream invertebrates respond to the type and spatial arrangement of patches. *Landsc. Ecol.* 15:563–76

Pan YD, Stevenson RJ, Hill BH, Kaufmann PR, Herlihy AT. 1999. Spatial patterns and ecological determinants of benthic algal assemblages in Mid-Atlantic streams, USA. *J. Phycol.* 35:460–68

Paul MJ, Meyer JL. 2001. Streams in the urban landscape. *Annu. Rev. Ecol. Syst.* 32:333–65

Perkins SJ. 1994. The shrinking Cedar River. Channel changes following flow regulation and bank armoring. In *Effects of Human-Induced Changes on Hydrologic Systems, Proc. Am. Water Resour. Assoc. 1994 Annu. Summer Symp,* Herndon, VA, 1994, ed. RA Marston, V Haffurther, pp. 649–58

Peterson JT, Kwak TJ. 1999. Modeling the effects of land use and climate change on riverine smallmouth bass. *Ecol. Appl.* 9:1391–404

Poff NL. 1997. Landscape filters and species traits: towards mechanistic understanding and prediction in stream ecology. *J. N. Am. Benthol. Soc.* 16:391–409

Poff NL, Allan DJ. 1995. Functional organization of stream fish assemblages in relation to hydrologic variability. *Ecology* 76:606–27

Poff NL, Allan DJ, Bain MB, Karr JR, Prestegaard KL, et al. 1997. The natural flow regime: a paradigm for conservation and restoration of riverine ecosystems. *BioScience* 47:769–84

Quinn JM. 2000. Effects of pastoral development. In *New Zealand Stream Invertebrates: Ecology and Implications for Management*, ed. KJ Collier, MJ Winterbourn, pp. 208–29. Christchurch, NZ: Caxton

Quinn JM, Brown PM, Boyce W, Mackay S, Taylor A, Fenton T. 2001. Riparian zone classification for management of stream water quality and ecosystem health. *J. Am. Water Resour. Assoc.* 37:1509–15

Quinn JM, Hickey CW. 1990. The magnitude of the effects of substrate particle size, recent flooding, and catchment development on benthic invertebrates in 88 New Zealand Rivers. *N. Z. J. Mar. Fresh.* 24:411–27

Regetz J. 2003. Landscape-level constraints on recruitment of Chinook salmon (*Oncorhynchus tshawytscha*) in the Columbia River Basin, USA. *Aquat. Conserv.* 13:35–49

Richards C, Haro RJ, Johnson LB, Host GE. 1997. Catchment- and reach-scale properties as indicators of macroinvertebrate species traits. *Freshw. Biol.* 37:219–30

Richards C, Johnson LB, Host GE. 1996. Landscape-scale influences on stream habitats and biota. *Can. J. Fish. Aquat. Sci.* 53: 295–311

Robinson CT, Tockner K, Ward JV. 2002. The fauna of dynamic riverine landscapes. *Freshw. Biol.* 47:661–77

Rolland RM. 2000. A review of chemically-induced alterations in thyroid and vitamin A status from field studies of wildlife and fish. *J. Wildl. Dis.* 36:615–35

Roth NE, Allan JD, Erickson DL. 1996. Landscape influences on stream biotic integrity assessed at multiple spatial scales. *Landsc. Ecol.* 11:141–56

Roy AH, Rosemond AD, Paul MJ, Leigh DS, Wallace JB. 2003. Stream macroinvertebrate response to catchment urbanisation (Georgia, USA). *Freshw. Biol.* 48:329–46

Schlosser IJ. 1985. Flow regime, juvenile abundance, and the assemblage structure of stream fishes. *Ecology* 66:1484–90

Schlosser IJ. 1991. Stream fish ecology: a landscape perspective. *BioScience* 41:704–12

Schulz R, Liess M. 1999. A field study of the effects of agriculturally derived insecticide input on stream macroinvertebrate dynamics. *Aquat. Toxicol.* 46:155–76

Scott MC, Helfman GS. 2001. Native invasions, homogenization, and the mismeasure of integrity of fish assemblages. *Fisheries* 26:6–15

Sedell JR, Froggatt JL. 1984. Importance of streamside forest to large rivers: the isolation of the Willamette River, Oregon, USA, from its floodplain by snagging and streamside forest removal. *Verh. Int. Ver. Theor. Ang. Limnol.* 22:1828–34

Skinner JA, Lewis KA, Bardon KS, Tucker P, Catt JA, Chambers BJ. 1997. An overview of the environmental impact of agriculture in the U.K. *J. Environ. Manag.* 50:111–28

Snelder TH, Biggs BJF. 2002. Multiscale river environment classification for water resources management. *J. Am. Water Resour. Assoc.* 38:1225–39

Sovell LA, Vondracek B, Frost JA, Mumford KG. 2000. Impacts of rotational grazing and riparian buffers on physicochemical and biological characteristics of southeastern Minnesota, USA, streams. *Environ. Manag.* 26:629–41

Sponseller RA, Benfield EF, Valett HM. 2001. Relationships between land use, spatial scale and stream macroinvertebrate communities. *Freshw. Biol.* 46:1409–24

Stauffer JC, Goldstein RM, Newman RM. 2000. Relationship of wooded riparian zones and runoff potential to fish community composition in agricultural streams. *Can. J. Fish. Aquat. Sci.* 57:307–16

Steedman RJ. 1988. Modification and assessment of an index of biotic integrity to quantify stream quality in southern Ontario. *Can. J. Fish. Aquat. Sci.* 45:492–501

Stepenuck KF, Crunkilton RL, Wang LZ. 2002. Impacts of urban land use on macroinvertebrate communities in southeastern Wisconsin streams. *J. Am. Water Resour. Assoc.* 38:1041–51

Stewart JS, Wang LZ, Lyons J, Horwatich JA, Bannerman R. 2001. Influences of watershed, riparian-corridor, and reach-scale characteristics on aquatic biota in agricultural watersheds. *J. Am. Water Resour. Assoc.* 37: 1475–87

Storey RG, Cowley DR. 1997. Recovery of three New Zealand rural streams as they pass through native forest remnants. *Hydrobiologia* 353:63–76

Strand M, Merritt RW. 1999. Impact of livestock grazing activities on stream insect communities and the riverine environment. *Am. Entomol.* 45:13–27

Strayer DL, Beighley RE, Thompson LC, Brooks S, Nilsson C, et al. 2003. Effects of land cover on stream ecosystems: roles of empirical models and scaling issues. *Ecosystems* 6:407–23

Sutherland AB, Meyer JL, Gardiner EP. 2002. Effects of land cover on sediment regime and fish assemblage structure in four southern Appalachian streams. *Freshw. Biol.* 47: 1791–805

Sweeney BW. 1993. Effects of streamside vegetation on macroinvertebrate communities of White Clay Creek in eastern North America. *Proc. Acad. Nat. Sci. Philadelphia,* Philadelphia, PA

Tockner K, Ward JV, Edwards PJ, Kollmann J. 2002. Riverine landscapes: an introduction. *Freshw. Biol.* 47:497–500

Tong STY, Chen WL. 2002. Modeling the relationship between land use and surface water quality. *J. Environ. Manag.* 66:377–93

Townsend CR. 1989. The patch dynamics concept of stream community ecology. *J. N. Am. Benthol. Soc.* 8:36–50

Townsend CR, Doledec S, Norris R, Peacock K, Arbuckle C. 2003. The influence of scale and geography on relationships between stream community composition and landscape variables: description and prediction. *Freshw. Biol.* 48:768–85

Townsend CR, Doledec S, Scarsbrook MR. 1997. Species traits in relation to temporal and spatial heterogeneity in streams: a test of habitat templet theory. *Freshw. Biol.* 37:367–87

Townsend CR, Hildrew AG. 1994. Species traits in relation to a habitat templet for river systems. *Freshw. Biol.* 31:265–76

Townsend CR, Riley RH. 1999. Assessment of river health: accounting for perturbation pathways in physical and ecological space. *Freshw. Biol.* 41:393–405

Turner MG, Gardner RH, O'Neill RV. 2001. *Landscape Ecology in Theory and Practice: Pattern and Process*. New York: Springer

Usseglio-Polatera P, Beisel JN. 2002. Longitudinal changes in macroinvertebrate assemblages in the Meuse River: anthropogenic effects versus natural change. *River Res. Appl.* 18:197–211

Usseglio-Polatera P, Bournaud M, Richoux P, Tachet H. 2000a. Biological and ecological traits of benthic freshwater macroinvertebrates: relationships and definition of groups with similar traits. *Freshw. Biol.* 43:175–205

Usseglio-Polatera P, Bournaud M, Richoux P, Tachet H. 2000b. Biomonitoring through biological traits of benthic macroinvertebrates: How to use species trait databases? *Hydrobiologia* 422:153–62

Vannote RL, Minshall WG, Cummins KW, Sedell JR, Cushing CE. 1980. The river continuum concept. *Can. J. Fish. Aquat. Sci.* 37: 130–37

Walser CA, Bart HL. 1999. Influence of agriculture on in-stream habitat and fish community structure in Piedmont watersheds of the Chattahoochee River System. *Ecol. Freshw. Fish* 8:237–46

Walsh CJ, Sharpe AK, Breen PF, Sonneman JA. 2001. Effects of urbanization on streams of the Melbourne region, Victoria, Australia. I. Benthic macroinvertebrate communities. *Freshw. Biol.* 46:535–51

Wang L, Lyons J, Rasmussen P, Seelbach P, et al. 2003. Watershed, reach, and riparian influences on stream fish assemblages in the Northern Lakes and Forest Ecoregion, U.S.A. *Can. J. Fish. Aquat. Sci.* 60:491–505

Wang L, Lyons J, Kanehl P. 2001. Impacts of urbanization on stream habitat and fish across multiple spatial scales. *Environ. Manag.* 28: 255–66

Wang L, Lyons J, Kanehl P. 2002. Effects of watershed best management practices on habitat and fish in Wisconsin streams. *J. Am. Water Resour. Assoc.* 38:663–80

Wang L, Lyons J, Kanehl P, Bannerman R, Emmons E. 2000. Watershed urbanization and changes in fish communities in southeastern Wisconsin streams. *J. Am. Water Resour. Assoc.* 36:1173–89

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

Ward JV. 1998. Riverine landscapes: biodiversity patterns, disturbance regimes, and aquatic conservation. *Biol. Conserv.* 83:269–78

Ward JV, Tockner K, Arscott DB, Claret C. 2002. Riverine landscape diversity. *Freshw. Biol.* 47:517–39

Waters TF. 1995. *Sediment in Streams*. Bethesda, MD: Am. Fish. Soc.

Watzin MC, McIntosh AW. 1999. Aquatic ecosystems in agricultural landscapes: a review of ecological indicators and achievable ecological outcomes. *J. Soil Water Conserv.* 54:636–44

Weller DE, Jordan TE, Correll DL. 1998. Heuristic models for material discharge from landscapes with riparian buffers. *Ecol. Appl.* 8:1156–69

Wiens JA. 1989. Spatial scaling in ecology. *Funct. Ecol.* 3:385–97

Wiens JA. 2002. Riverine landscapes: taking landscape ecology into the water. *Freshw. Biol.* 47:501–15

Wissmar RC, Beschta RL. 1998. Restoration and management of riparian ecosystems: a catchment perspective. *Freshw. Biol.* 40:571–85

Wood PJ, Armitage PD. 1997. Biological effects of fine sediment in the lotic environment. *Environ. Manag.* 21:203–17

Woodward DF, Goldstein JN, Farag AM, Brumbaugh WG. 1997. Cutthroat trout avoidance of metals and conditions characteristic of a mining waste site: Coeur d'Alene River, Idaho. *Trans. Am. Fish. Soc.* 126:699–706

Wright JF. 1995. Development and use of a system for predicting macroinvertebrates in flowing waters. *Aust. J. Ecol.* 20:181–97

Yuan LL, Norton SB. 2003. Comparing responses of macroinvertebrate metrics to increasing stress. *J. N. Am. Benthol. Soc.* 22: 308–22