FOREST COVER, IMPERVIOUS-SURFACE AREA, AND THE MITIGATION OF STORMWATER IMPACTS[1]

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ABSTRACT: For 20 years, King County, Washington, has implemented progressively more demanding structural and nonstructural strategies in an attempt to protect aquatic resources and declining salmon populations from the cumulative effects of urbanization. This history holds lessons for planners, engineers, and resource managers throughout other urbanizing regions. Detention ponds, even with increasingly restrictive designs, have still proven inadequate to prevent channel erosion. Costly structural retrofits of urbanized watersheds can mitigate certain problems, such as flooding or erosion, but cannot restore the predevelopment flow regime or habitat conditions. Widespread conversion of forest to pasture or grass in rural areas, generally unregulated by most jurisdictions, degrades aquatic systems even when watershed imperviousness remains low. Preservation of aquatic resources in developing areas will require integrated mitigation, which must including impervious-surface limits, forest-retention policies, stormwater detention, riparian-buffer maintenance, and protection of wetlands and unstable slopes. New management goals are needed for those watersheds whose existing development precludes significant ecosystem recovery; the same goals cannot be achieved in both developed and undeveloped watersheds.
(KEY TERMS: urbanization; stormwater; BMP; land use planning; watershed management; urban water management.)

INTRODUCTION

For decades, watershed urbanization has been known to harm aquatic systems. Although the problem has been long articulated, solutions have been elusive because of the complexity of the problem, the evolution of still-imperfect analytical tools, and socioeconomic forces with different and often incompatible interests. King County, Washington, has been a recognized leader in the effort to analyze and to reduce the consequences of urban development, but even in this jurisdiction the path toward aquatic resource protection has been marked by well-intentioned but ultimately mistaken approaches, compromises with other agency goals that thwart complete success, and imperfect implementation of adopted policies and plans. This experience demonstrates the difficulty of meeting urban and suburban water-quality and aquatic-resource protection goals in the face of competing social priorities and variable political resolve on environmental issues that require sustained, long-term strategies to achieve progress.

King County provides a useful case study for resource managers in urbanizing regions across the country. It covers about 5,600 square kilometers with a population of 1.7 million people, the twelfth most populous county in the United States. Its western boundary is Puget Sound and its eastern boundary is the crest of the Cascade Range. It contains all or most of three major river basins, two large natural lakes, and numerous small rivers and streams (Figure 1). The streams and lakes support all species of anadromous Pacific salmon and resident trout. Land uses include urban, industrial, suburban, agriculture, rural, commercial timber production, and National Forest. Cities include Seattle, Bellevue, Renton, and Redmond; population growth has been explosive over the last 20 years.

Recent Endangered Species Act (ESA) listings of Puget Sound chinook and bull trout, and the potential for more salmonid listings, have brought new scrutiny to all aspects of watershed protection and urbanization-mitigation efforts in King County and

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the surrounding region. Such increased attention is forcing improved articulation of the goals, the means, and the justification for mitigating the effects of urban development. It also has highlighted the failure of most stormwater mitigation efforts, not only in the Pacific Northwest but also across the country, where well-publicized successes are overshadowed by progressive degradation of once-healthy stream systems. This degradation has continued, despite sincere but ineffectual efforts via structural "Best Management Practices" (BMPs), particularly detention ponds, buffer regulations, and rural zoning.

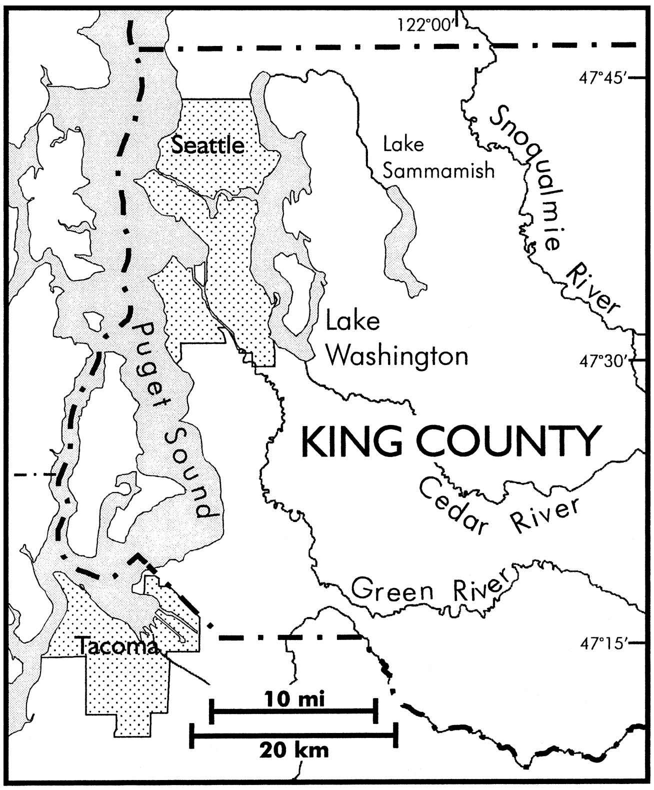

Figure 1. Location of King County, Washington. Most urban and suburban development here is occurring in the region between Puget Sound and the Snoqualmie River.

Our purpose here is to diagnose what has gone wrong with these structural and regulatory approaches, so that others can think more creatively and productively about potentially more successful strategies, and to suggest preliminary solutions of our own. Our approach has four elements: (1) to review some empirical relationships between watershed conditions and stream conditions; (2) to review the history of surface-water management in King County as it relates to the analysis and mitigation of urban development; (3) to evaluate the basis for regulating watershed land use, rather than building structural BMPs, to minimize the downstream consequences of urbanization; and (4) to recommend an integrated stormwater management strategy based on King County's experience of the past decade. We have no panaceas, however. If the problems were easily solved, they would have been so many years ago.

This paper focuses on changes in runoff and stream flow because they are ubiquitous in urbanizing basins and cause often dramatic changes in flooding, erosion, sediment transport, and ultimately channel morphology. Hydrologic change also influences the whole range of environmental features that affect aquatic biota – flow regime, aquatic habitat structure, water quality, biotic interactions, and food sources (Karr, 1991). Yet runoff and stream-flow regime, while important, are by no means the only drivers of aquatic health. Consequently, there should be no illusion that *just* addressing hydrologic conditions will necessarily "fix" or "protect" an urban stream.

Modifications of the land surface during urbanization produce changes in both the magnitude and the type of runoff processes. In the Pacific Northwest, the fundamental hydrologic effect of urban development is the loss of water storage in the soil column. This may occur because the soil is compacted or stripped during the course of development, or because impervious surfaces convert what was once subsurface runoff to Horton overland flow. In either situation, the precipitation over a small watershed reaches the stream channel with a typical delay of just a few minutes, instead of what had been a lag of hours, days, or even weeks. The result is a dramatic chang in flow patterns in the downstream channel, with the largest flood peaks doubled or more and more frequent storm discharges increased by as much as ten-fold (Figure 2).

EMPIRICAL RELATIONSHIPS BETWEEN WATERSHED CONDITIONS AND STREAM CONDITIONS

Correlations between watershed development and aquatic-system conditions have been investigated for over two decades. Klein (1979) published the first such study, where he reported a rapid decline in biotic diversity where watershed imperviousness exceeded 10 percent. Steedman (1988) believed that his data showed the consequences of both impervious cover and forest cover on instream biological conditions. Later studies, mainly unpublished but covering a large number of methods and researchers, were compiled by Schueler (1994). Since that time, additional work on this subject has been done by a variety of Pacific Northwest researchers, including May (1996), Booth and Jackson (1997), and Morley (2000) (Figures 3, 4, and 5).

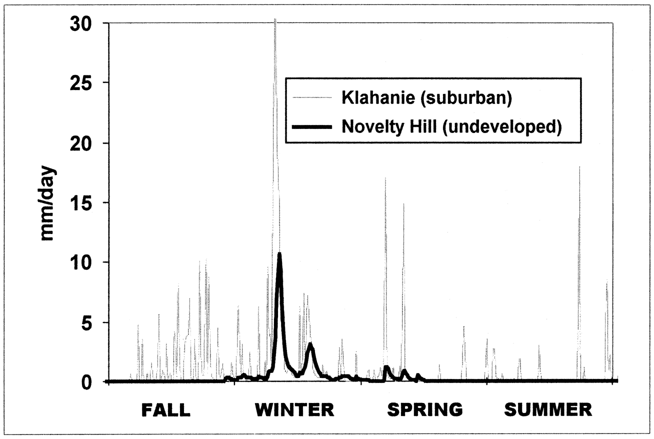

Figure 2. One year's measured discharges for a suburban (Klahanie) and an undeveloped (Novelty Hill) watershed, normalized by basin area (data from Burges *et al.,* 1998).

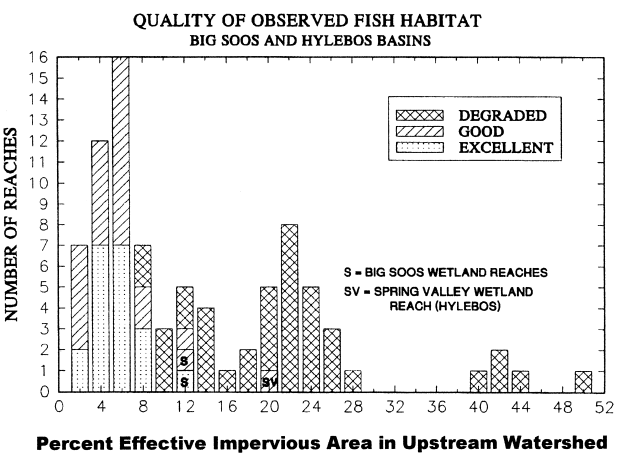

Figure 3. Observed fish habitat quality as a function of effective impervious area in the contributing watershed, based on more than 80 individually inventoried channel segments in south King County (from Booth and Jackson, 1997; data from King County, 1990a, 1990c). "EXCELLENT" reaches show little or no habitat degradation; "GOOD" reaches show some damage to habitat but still maintain good biological function; and "DEGRADED" reaches contain aquatic habitat that has been clearly and extensively damaged, typically from bank erosion, channel incision, and sedimentation.

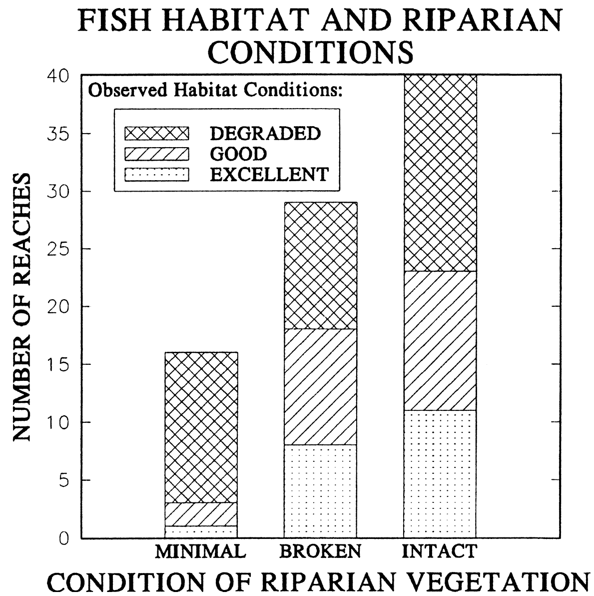

Figure 4. Relationship between riparian vegetation and instream conditions, using the same sites and criteria as for Figure 3. A relatively intact riparian corridor is clearly necessary, but not sufficient, for high quality habitat.

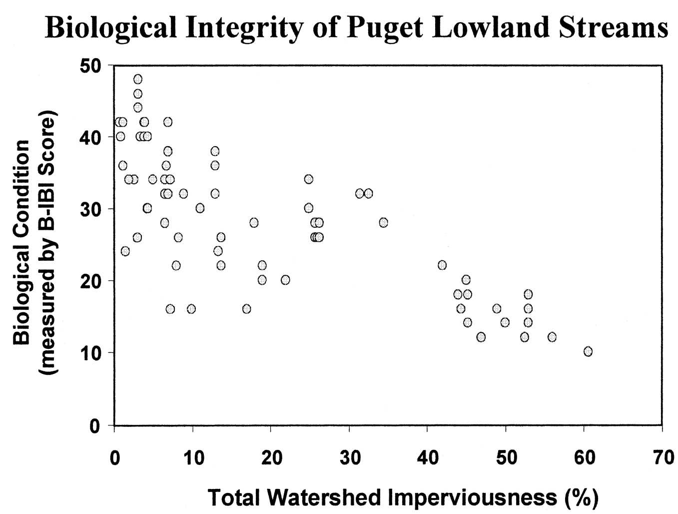

Figure 5. Compilation of biological data on Puget Lowland watersheds, reported by Kleindl (1995), May (1996), and Morley (2000). The pattern of progressive decline with increasing imperviousness in the upstream watershed is evident only in the upper bound of the data; significant degradation can occur at any level of human disturbance (at least as measured by impervious cover).

These data have several overall implications:

• "Imperviousness," although an imperfect measure of human influence, is clearly associated with stream-system decline. A wide *range* of stream conditions, however, can be associated with any given level of imperviousness, particularly at lower levels of development.

• "Thresholds of effect," articulated in some of the earlier literature (e.g., Klein, 1979; Booth and Reinelt, 1993) exist largely as a function of measurement (im)precision, not an intrinsic characteristic of the system being measured. Crude evaluation tools require that large changes accrue before they can be detected, but lower levels of development may still have consequences that can be revealed by other, more sensitive methods. In particular, biological indicators (e.g., Figure 5) demonstrate a continuum of effects, not a threshold response, resulting from human disturbance.

MITIGATION OF NEW DEVELOPMENT: THE KING COUNTY, WASHINGTON, EXPERIENCE

Hydrologic Mitigation Through Structural Means

As a consequence of the urban-induced runoff changes that cause flooding, erosion, and habitat damage, jurisdictions have long required some degree of stormwater mitigation for new developments. The most common approach has been to reduce flows through the use of detention ponds, which are intended to capture and detain stormwater runoff from developed areas. These ponds can be designed to either of two levels of performance, depending on the desired balance between achieving downstream protection and the cost of providing that protection. A *peak standard*, the classic (and least costly) goal of detention facilities, seeks to maintain post-development peak discharges at their predevelopment levels. Even if this goal is successfully achieved the aggregate duration that such flows occupy the channel must increase because the overall volume of runoff is greater.

In contrast, a *duration standard* seeks to maintain the post-development duration of a wide range of peak discharges at predevelopment levels. Yet unless runoff is infiltrated, the total *volume* of runoff must still increase in the post-development condition. Thus durations cannot be matched for all discharges because this "excess" water must also be released. Duration standards seek to avoid potential disruption to the downstream channels by choosing a "threshold discharge," below which sediment transport in the receiving channel is presumed not to occur and so post-development flow durations can be increased without concern. This choice can be made by site-specific, but rather expensive, analysis based on stream hydraulics and sediment size (Buffington and Montgomery, 1997) or can be applied as a "generic" standard based on predevelopment discharges.

The first efforts at runoff mitigation sought to reduce peak flows, reflecting the traditional focus on flood reduction. Well over 100 years ago, the fundamental predicting equation of runoff used in these early mitigation efforts was developed (Mulvany, 1851). The Rational Runoff Formula related the runoff rate to the simple product of the rate of rainfall, the basin area, and the *runoff coefficient,* a number equal to the fraction of the rain falling on a basin that presumably contributes to the flood peak. This formula was used by King County in the Pacific Northwest region's first surface-water design manual (King County, 1979). Unfortunately, it tended to overestimate predevelopment flows, which led to the construction of grossly undersized detention ponds that had little or no benefit in preventing downstream flooding (Booth and Jackson, 1997). Ponds designed with the Rational method had such high release rates that they rarely backed up water during storms.

The subsequent edition of King County's design manual (King County, 1990b) substituted the Soil Conservation Service's (SCS) curve-number methodology for the Rational equation. This was a dramatic, and costly, change on several fronts: (1) it nominally allowed for closer matching of watershed conditions by the modeling; (2) it generally yielded a requirement for larger detention ponds; and (3) it necessitated significant additional training in hydrologic-modeling skills for local engineers doing drainage-design work. Although it was an improvement over the Rational method, the SCS method still contained fundamental flaws that resulted in detention ponds that did not meet desired performance criteria. In this method, runoff from individual 24-hour design storm events was used to test and adjust pond designs, and ponds were assumed to be empty at the beginning of a storm. Yet this is rarely the case during (commonly sequential) wet-season storms. SCS curve-number hydrology also commonly overestimated predevelopment flows, a tendency sometimes exacerbated by design engineers who manipulated the time of concentration and curve number to reduce the size of the pond on their client's behalf. Furthermore, the SCS methodology was still a "peak standard" that ignored any problems associated with increased flow durations. Continuous flow modeling revealed that the ponds designed with the SCS method would not achieve the stated protection goals (Barker *et al.,* 1991). Although convincing the land developers and their engineers of these problems has proven difficult, the county's 1998 version of the Design Manual did incorporate a regionally calibrated continuous flow model for designing stormwater facilities (King County, 1998; Jackson *et al.,* 2001).

The practice of seeking duration control for new developments was introduced through King County's Basin Planning Program in the late 1980s. The goal of this standard is to match pre- and post-development flow durations for all discharges above a chosen threshold. Hydrologic analysis using a more advanced (albeit still imperfect) hydrologic model, HSPF (Hydrologic Simulation Program-Fortran) (Bicknell *et al.,* 1997), could predict the detention needed to achieve this goal (Jackson *et al.,* 2001).

From the outset, this approach has been controversial for several reasons:

1. The required ponds are larger, often dramatically so, than required by previous design methods.
2. The method requires a threshold discharge, below which durations will increase dramatically, but how to choose that discharge is not immediately obvious or without dispute.
3. The analytic tool (HSPF) used to establish the standard is not as widely used as the Rational or SCS method, and so appeared less transparently justifiable to many practitioners. For example, as part of the Bear Creek Basin Plan (King County, 1990d) a surrogate approach that involved an intentional "misapplication" of the SCS method was proposed to achieve the same objective without requiring the ability to run HSPF.
4. Few (and initially, no) ponds were actually constructed under this standard, and so empirical evidence for their effectiveness (or lack thereof) is sparse.

Despite these shortcomings, these standards reflected the best understanding of hydrologic conditions in urban streams and so have been part of Basin Plan-recommended detention standards in King County since the early 1990s [and incorporated into more recent updates (1998) of the design manual]. Yet several issues remain unanswered, even with the current status of implementation:

"Threshold" Discharge. As noted above, there is a presumed threshold discharge below which there are "no effects" of flow-duration increase. This may be defensible, at best, with regard to sediment transport in gravel-bed streams. A true "threshold of no effects" is certainly *not* correct for sediment transport in sand-bedded streams (uncommon but not unknown in the region); some bed material moves at almost any discharge. In addition, there has been no evaluation of any other effects (either physical or biological) of extended low-flow durations.

Point Discharge. These analyses ignore the consequences of converting what was once spatially distributed subsurface runoff into a point discharge at a surface-water outfall, because there are no analytic tools to assess those consequences. Field examples, however, demonstrate that the consequences of point discharges can include locally severe erosion and disruption of riparian vegetation and instream habitat (e.g., Booth, 1990).

Ground Water. Any analysis of flow durations will not address changes to ground water recharge or discharge, because no constructed detention ponds, even the largest designed under this standard, can delay wintertime rainfall sufficiently for it to become summertime runoff. Yet exactly this magnitude of delay *does* occur under predevelopment conditions, because far more of the precipitation is stored as ground water.

Individual Storm Hydrographs. The flow-duration design, by definition, assures that the fractional time of a given discharge's exceedence remains unchanged over an extended climate record (nearly 50 years, in the case of King County), but there is no attempt (or ability) to construct detention ponds that match durations for specific storm events or even an entire storm season. Thus the *aggregate* flow-duration spectrum may be unchanged, but the timing and brevity of any single storm hydrograph may be quite different from the undisturbed condition.

Des Moines Creek, a small urban system, demonstrates these difficulties in accomplishing the hydrologic restoration in an urban stream. Since the 1940s, widespread conversion of forests and pastures has occurred to accommodate Seattle-Tacoma International Airport and other commercial and residential uses. Within the Creek's 14 km^2 watershed, total impervious area was raised approximately 50 percent, wetlands were filled, some of the stream headwaters were piped, and storm runoff to the remaining natural drainage system was discharged with minimal detention. As a result, increased magnitude, frequency, and duration of peak flows raised flow velocities, destabilized the stream channel, eroded spawning gravels, degraded fish habitat, and caused flooding of park facilities near the mouth of the stream. Additionally, summer base flows and water quality declined in the Creek.

By the 1990s, the public and local government resolved to develop and implement a basin plan to solve these problems and restore the creek. However, the challenges faced by the technical and policy teams were formidable (Des Moines Creek Basin Committee, 1997). Any solution to existing problems also needed to accommodate additional future development within the watershed that would raise total impervious area from approximately 50 percent to 65 percent of the total drainage area and to have a cost acceptable to the participating jurisdictions.

Hydrologic modeling was used to evaluate feasible combinations of on-site detention ponds, regional flow bypasses, and regional detention ponds to reduce storm-flow energy in the creek. For $6 million, covering a range of feasible options, very large reductions in flows and flow energy compared to 1990s conditions could be achieved. Yet none of these options could restore storm flows to pristine conditions. The preferred alternative combined peak control with on-site detention ponds, regional detention, and a preexisting pipeline to bypass peak stormwater flows. This alternative provides dramatic flow-duration improvement over current conditions (Figure 6a), but daily flows in the stream do not even begin to approximate pristine conditions, despite a capital cost of nearly $5,000 per watershed hectare (almost $2,000/acre) (Figure 6b).

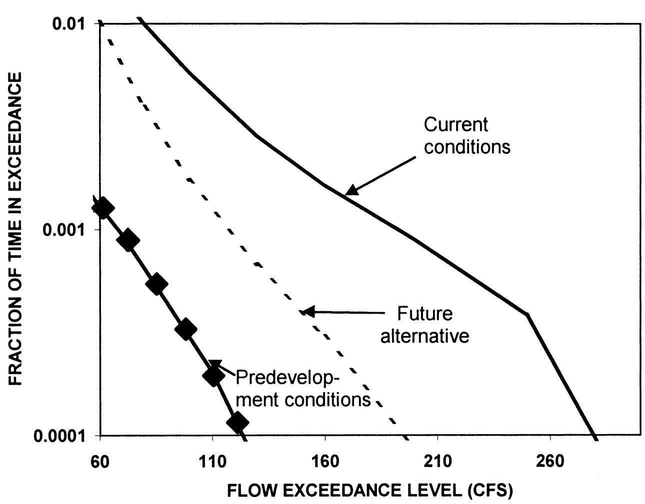

Figure 6a. HSPF-modeled flow-duration curve for Des Moines Creek, displaying dramatic improvement in future flow durations relative to current. Analysis assumes projected land-use changes and construction of proposed detention ponds and bypass pipeline (from Des Moines Basin Committee, 1997).

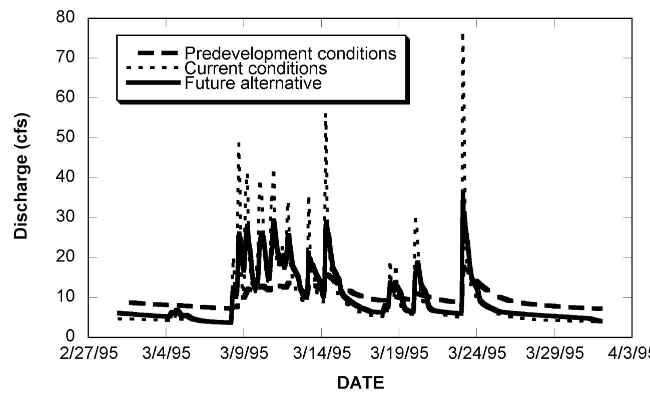

Figure 6b. One month's hydrographs for Des Moines Creek: current flows, predevelopment (i.e., forested) flows, and those under the anticipated future (mitigated) alternative. Note that although the flow-duration curves (Figure 6a) suggest that the future alternative is about mid-way between current and predevelopment conditions, the future hydrograph shows flashy discharge and low base flows much more like current (urban) conditions than those of predevelopment time.

Hydrologic Restoration Through Watershed Planning

Realizing that on-site drainage controls alone were insufficient to achieve the goals of either stormwater management or resource protection, King County initiated an interdisciplinary watershed planning program in the mid 1980s, with the goal of solving and preventing flooding, water-quality, and habitat problems within the rapidly-urbanizing western part of the county. This "basin planning process" involved a two step approach:

1. A detailed assessment of basin conditions that included inventories of point and nonpoint pollution sources, characterization of channel habitat and fish communities, mapping existing and anticipated land uses, identifying and characterizing flooding and channel erosion problems, and modeling stream flows under various development scenarios using HSPF.
2. Development of solutions that combined constructed projects, drainage and zoning regulations, and public education programs.

One finding of the early plans was that aquatic resources had been degraded by low-density rural development (e.g., one dwelling unit per five acres) (King County, 1990a, 1990d). Although this density of development generally did not create much imperviousness, the amount of forest clearing to create large lawns, pastures, or hobby farms could easily reach 60 percent of the landscape, with significant effects on watershed flow regime. Furthermore, many rural landowners were inclined to "manage" the streams on their property. This might include riparian forest clearing, removing woody debris from the channel, and hardening stream banks to protect property. Rural zoning, in and of itself, does not necessarily protect aquatic resources.

The failure of simple land-use controls (i.e., zoning) to protect aquatic resources led to the need for objective criterion for "acceptable" hydrologic performance that might protect stream channels. This "stream-protection" criterion was taken directly from previous empirical assessments of channel stability and bank erosion, which in turn had been generated from observations made in the late 1980s and early 1990s while working on the past and current basin plans (and subsequently published in Booth and Jackson, 1997) (Figure 7). These data showed that two linked thresholds apparently marked a transition of the visible channel form from "stable" to "unstable" (see also Henshaw and Booth, 2000). One was the measure discussed previously – where effective impervious area in the contributing watershed had exceeded 10 percent, readily observed physical degradation of the channel was ubiquitous. The other was based on hydrologic analyses of those same contributing watersheds – almost without exception, the same observed transition from "stable" to "unstable" channels was marked by the equality of the ten-year forested (i.e., predevelopment) discharge ($Q_{10\text{-for}}$) and the two-year current discharge ($Q_{2\text{-urban}}$). There was, and is, no theoretical basis for these particular outcomes – they are simply empirical results, remarkable in their consistency across western Washington and quite possibly recognizable in other regions of the country as well (Schueler, 1994).

Although these data compose a robust set of observations, spanning a wide variety of streams with remarkably consistent results, they also carry two limitations. First, the absence of observed instability does not guarantee an absence of *any* effects. The second limitation is more vexing: these data were collected on watersheds without much, if any, effective stormwater detention. Had larger and more effective ponds been present, would the observed impacts been reduced? Recent investigations by Maxted and Shaver (1999) suggest virtually no improvement in stream conditions from typical detention ponds. Even if they could be designed to be hydrologically effective, ponds cannot avoid other key problems such as disruption of storm flow patterns, increased winter storm volumes, or declining base flows.

Notwithstanding these limitations (i.e., potentially unrecognized degradation and potentially effective detention ponds), the Issaquah Creek Basin Plan (King County, 1994) used the "threshold" criteria for stream-channel stability suggested by Figure 7 to evaluate the likely consequences of model predictions of post-development runoff conditions. These initial assessments, presuming basinwide application of the mitigation tools that were then "accepted practice" (i.e., exemption of rural-zoned developments from detention requirements, and SCS-based hydrologic designs for the rest), produced results that were inconsistent with the goals of the basin plan – to protect aquatic habitat and to resolve existing and potential future flooding problems. The empirical hydrologic criterion for channel instability ($Q_{2\text{-urban}} > Q_{10\text{-for}}$) was exceeded pervasively throughout the watershed under all future development scenarios.

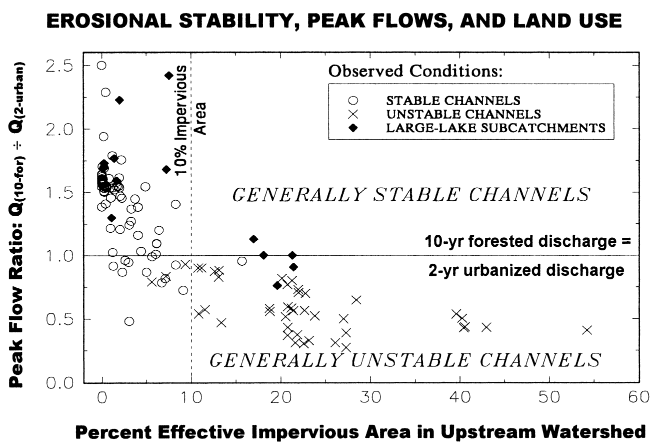

Figure 7. Observed stable ("O") and unstable ("X") channels, plotted by percent effective impervious area (EIA) in the upstream watershed (horizontal scale) and ratio of modeled ten-year forested and two-year urbanized (i.e., current) discharges (vertical scale). "Stable channels" consistently meet the apparent thresholds of either {EIA ≤ 10 percent} or {$Q_{2\text{-urban}} \leq Q_{10\text{-for}}$}, except for the few catchments containing large lakes (from Booth and Jackson, 1997).

As a consequence of these results, the Issaquah plan evaluated a variety of alternative rural development scenarios (Appendix G of King County, 1994). The analyses found that with 65 percent forest retention in a nominal five-acre zone (i.e., 20 houses per 100 acres, but clustered on the nonforested 35 percent of the land area), the criterion of keeping the two-year developed discharge below the ten-year forested discharge could be just met on glacial till soils (the most common type in King County). Greater amounts of cleared land resulted in two-year developed discharges that exceeded ten-year forested discharges, even though the amount of effective impervious area was well under 10 percent. The analysis noted that development on highly pervious glacial outwash soils (the other, but much less common, soil type used for hydrologic modeling) failed the criterion at virtually any level of forest retention, because so little runoff occurs there naturally that almost any amount of imperviousness produces proportionally large peak-flow increases. The analysis also found that in rural areas, forest clearing and conversion to suburban vegetation (mainly lawns) was far more significant in determining peak discharge increases than the small increases in impervious area typical of low-density development (Figure 8). As a result, forest retention has been adopted as an alternative to detention for rural plats and short plats in the latest update to the Stormwater Design Manual.

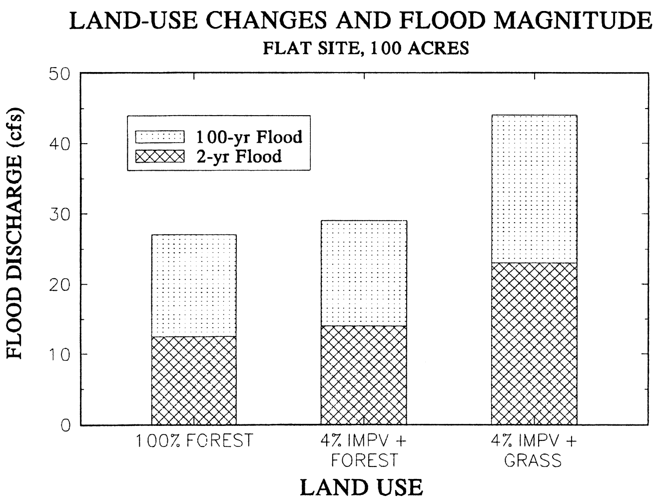

Figure 8. HSPF-modeled increases in two-year and 100-year discharges that result from forest conversion on moderately sloping till soils. Four percent (effective) imperviousness, a typical value for five-acre residential densities, shows particularly significant hydrologic changes only when accompanied by forest clearing.

THE BASIS FOR REGULATING IMPERVIOUS AREA AND CLEARING

In the realm of physical channel conditions, the data collected from field observations have consistently shown remarkably clear trends in aquatic-system degradation. In this region, approximately 10 percent effective impervious area in a watershed typically yields demonstrable degradation, some aspects of which are surely irreversible. Although early observations were not sensitive enough to show significant degradation at even lower levels of urban development, the basin plans of the early 1990s recognized that such damage was almost certainly occurring. More recently, biological data (e.g., Morley, 2000) have demonstrated the anticipated consequences at these lower levels of human disturbances.

Less empirical data have been collected on the direct correlation between forest cover and stream conditions than for watershed imperviousness and stream conditions. In general, the "evidence" has been based on the observed correlation of channel instability to the modeled hydrologic condition of $Q_{2\text{-urban}}$ greater than $Q_{10\text{-for}}$, coupled with hydrologic analyses that have explored the relationship between forest-cover reduction and peak-flow increases. The first such analyses, for the Issaquah Creek Basin Plan, made a variety of assumptions about "typical" watershed characteristics in that basin and found that 65 percent forest cover with 4 percent effective impervious area closely approached the condition of $Q_{2\text{-urban}} = Q_{10\text{-for}}$. Using more generalized model parameters and a range of effective impervious areas typical of rural areas, 65 percent forest cover is a plausible, but by no means definitive, value for meeting the presumed "stability criterion" of $Q_{2\text{-urban}}$ less than $Q_{10\text{-for}}$ in rural-zoned watersheds on moderately (5 to 15 percent) sloping till soils (Figure 9). The analysis summarized in Figure 9 assumes no on-site detention facilities are present because they are often technically (and politically) infeasible in low-density rural areas. Other soils (particularly more infiltrative ones) may yield much greater hydrologic response with even lesser amounts of clearing.

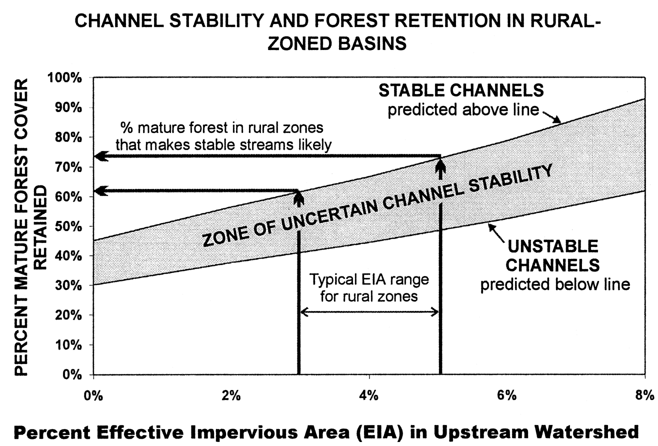

Figure 9. Conditions of forest cover and impervious area in an HSPF-modeled watershed, with moderate slopes and till soils, relative to the channel-stability criterion $Q_{2\text{-urban}} = Q_{10\text{-for}}$. The range of effective impervious areas (EIA = 3 to 5 percent) reflects variation in rural land cover conditions; the "zone of uncertain channel stability" reflects uncertainty in the hydrologic parameters.

Hydrological analyses suggest that maintaining forest cover is more important than limiting impervious-area percentages, at least at rural residential densities where zoning effectively limits the range of EIA between 2 and 6 percent of the gross development area. Absent clearing limitations, however, forest cover will range between 5 and about 85 percent. Consequently, even if both types of land cover control (i.e., forest retention and EIA limitation) are critical to protect stream conditions, current land-use practices suggest that mandating retention of forest cover is the more pressing regulatory need in rural areas. Degraded watersheds, with less than 10 percent EIA and less than 65 percent forest cover, are common ("cleared rural"); in contrast, we have found *no* watersheds with more than 10 percent EIA that have also retained at least 65 percent forest cover ("forested urban") (Figure 10).

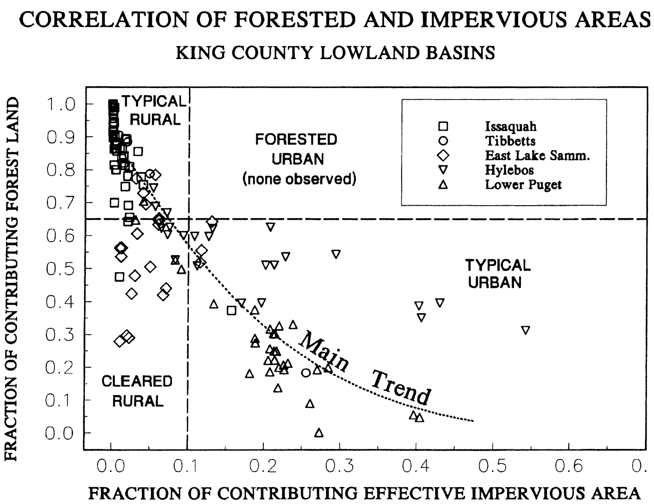

Figure 10. Land cover data from individual subcatchments within five King County watersheds, compiled from Basin Plan land-cover data (King County, 1990c, 1990e, 1991). At 65-percent forest retention, EIA ≤ 10 percent in all cases, yet with EIA < 10 percent, substantial clearing is still commonly observed.

The apparent correlations between stream stability and both impervious-area and forest-cover percentages present a quandary for watershed managers. On the one hand, these correlations point to a tangible, defensible criteria for achieving a specific management objective, namely "stable stream channels." On the other hand, this objective, however worthy, *still* allows the possibility of serious and significant aquatic-system degradation – and as development is allowed to approach these clearing and imperviousness criteria, degradation is virtually guaranteed. The thresholds implied by these data are simply the "wrong" type on which to base genuine resource protection. They do not separate a condition of "no impact" from that of "some impact;" instead, they separate the condition of "some impact" from that of "gross and easily perceived impact." Hydrologically and biologically, there are no truly negligible amounts of clearing or watershed imperviousness (Morley, 2000), even though our perception of, and our tolerance for, many of the associated changes in downstream channels appear to undergo a relatively abrupt transition. Almost every increment of cleared land, and of constructed pavement, is likely to result in some degree of resource degradation of loss. The decision of how much is "acceptable" is thus as much a social decision as a hydrologic one.

These conditions also emphasize the need to develop new approaches to mitigate the consequences of watershed urbanization on streams. If urban and suburban watersheds cannot hydrologically mimic forested ones, no matter how large their associated detention ponds, then reducing the coverage of effective impervious area or the extent of urban development itself is an inescapable consequence of the present desire to "restore" urban watercourses. If those necessary reductions run counter to other, even more pressing social goals, most notably those to accommodate additional population growth, then our goals for aquatic-resource conservation need to be modified in urban areas. By not acknowledging the need for such tradeoffs, opportunities to discover the most rational and effective strategy for protecting the condition of once-natural aquatic systems continue to be lost.

CONCLUSIONS AND MANAGEMENT IMPLICATIONS

Land development that eliminates hydrologically mature forest cover and undisturbed soil can result in significant changes to urban stream flow regimes and, in turn, to the physical stability of stream channels. These changes are manifested in altered stream flow patterns with higher volumes of storm flow, leading to accelerated channel erosion and habitat simplification. Even with stormwater detention ponds, seasonal and stormflow patterns are substantially different from those to which native biota have adapted. These hydrologic changes cannot be completely mitigated with structural measures. Although factors other than hydrologic change (e.g., water chemistry, riparian buffers) can undoubtedly affect the magnitude of urban impacts, the breadth of the existing data suggest that improvements in these other factors can never fully mitigate the hydrologic consequences of overly intense urban development. Under typical rural land uses, the magnitude of observed forest-cover losses affects watershed flow regime as much as, or more than, associated increases in impervious area.

The goals of stormwater detention have become progressively more ambitious as the consequences of urban-altered flow regime have become better recognized and understood. Even the largest detention ponds, however, are limited in their ability to mitigate all aspects of hydrologic change. Twenty years of empirical data display a good correlation between readily observed damage to channels and modeled changes in flow regime that correspond to loss of about one-third of the forest cover in a "typical" western Washington watershed. A similar degree of observed damage also correlates to a level of watershed effective imperviousness (EIA) of about ten percent.

Field observations and hydrologic modeling showed that the watershed plans of the early- to mid-1990s could only hope to meet plan-stipulated goals for resource protection by imposing clearing and impervious-area restrictions. The most commonly chosen thresholds, maximum 10 percent EIA and minimum 65 percent forest cover, mark an observed transition in the downstream channels from minimally to severely degraded stream conditions. At lower levels of human disturbance, aquatic-system damage may range from slight to severe but is nearly everywhere recognizable with appropriate monitoring tools. Not every watershed responds equally to a given level of human disturbance, but some degree of measurable resource degradation can be seen at virtually any level of urban development. The apparent "threshold" of observed stream-channel stability has no correlative in measured biological conditions; for any given watershed, additional development tends to produce additional aquatic-system degradation. However, these impervious and forest-retention percentages have proven to be attractive regulatory thresholds and are being advocated by the National Marine Fisheries Service as necessary conditions for mandated protection of rural areas under the Endangered Species Act.

Development that minimizes the damage to aquatic resources cannot rely on structural BMP's, because there is no evidence that they can mitigate any but the most egregious consequences of urbanization. Instead, control of watershed land-cover changes, including limits to both imperviousness and clearing, must be incorporated (see also Horner and May, 1999). We anticipate needing all of the following elements to maintain the possibility of effective protection:

- clustered developments that protect half or more of the forest cover, preferentially in headwater areas and around streams and wetlands to maintain intact riparian buffers;

- a maximum of 20 percent total impervious area, and substantially less effective impervious area through the widespread reinfiltration of stormwater (Konrad and Burges, 2001);

- on-site detention, realistically designed to control flow durations (not just peaks);

- riparian buffer and wetland protection zones that minimize road and utility crossings as well as overall clearing; and

- no construction on steep or unstable slopes.

Past experience suggests that each of these factors are important. However, we still lack empirical data on the response of aquatic resources to such "well-designed" developments. Therefore, these recommendations are based only on extrapolations, model results, and judgment; they are tentative at best. Where development has already occurred, these conditions clearly cannot be met and different management objectives are inescapable: many, perhaps all, streams in already-urban areas cannot be truly protected or restored, and a significant degree of probably irreversible stream degradation is unavoidable in these settings.

We can recognize why streams nominally protected under past drainage regulations have experienced severe degradation, we can articulate the kinds of development styles and strategies that should minimize new examples of degraded streams, and we can recognize the role of watershed land-cover regulation in minimizing the consequences of new development, but we cannot find any basis to expect that the full range of hydrological and ecological conditions can be replaced in a now-degraded urban channel. The key tasks facing watershed managers, and the public that can support or impede their efforts, are therefore: (1) to identify those watersheds where existing low urbanization and associated high-quality stream conditions that warrant the kinds of development conditions that may protect much of the existing quality of these systems; and (2) to develop a new set of management goals for those watersheds whose surrounding development precludes significant ecosystem recovery. Following the same strategy in *all* watersheds, developed and undeveloped alike, simply makes no sense.

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INFLUENCES OF WATERSHED, RIPARIAN-CORRIDOR, AND REACH-SCALE CHARACTERISTICS ON AQUATIC BIOTA IN AGRICULTURAL WATERSHEDS[1]

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ABSTRACT: Multivariate analyses and correlations revealed strong relations between watershed and riparian-corridor land cover, and reach-scale habitat versus fish and macroinvertebrate assemblages in 38 warmwater streams in eastern Wisconsin. Watersheds were dominated by agricultural use, and ranged in size from 9 to 71 km^2. Watershed land cover was summarized from satellite-derived data for the area outside a 30-m buffer. Riparian land cover was interpreted from digital orthophotos within 10-, 10- to 20-, and 20- to 30-m buffers. Reach-scale habitat, fish, and macroinvertebrates were collected in 1998 and biotic indices calculated. Correlations between land cover, habitat, and stream-quality indicators revealed significant relations at the watershed, riparian-corridor, and reach scales. At the watershed scale, fish diversity, intolerant fish and EPT species increased, and Hilsenhoff biotic index (HBI) decreased as percent forest increased. At the riparian-corridor scale, EPT species decreased and HBI increased as riparian vegetation became more fragmented. For the reach, EPT species decreased with embeddedness. Multivariate analyses further indicated that riparian (percent agriculture, grassland, urban and forest, and fragmentation of vegetation), watershed (percent forest) and reach-scale characteristics (embeddedness) were the most important variables influencing fish (IBI, density, diversity, number, and percent tolerant and insectivorous species) and macroinvertebrate (HBI and EPT) communities.
(KEY TERMS: riparian; aquatic biota; agriculture; watershed; land cover; biotic integrity.)

INTRODUCTION

Numerous studies have indicated that land use/cover can play an important role in determining stream water quality by influencing factors that control runoff, sediments, nutrients, flow, water temperature and channel morphology (Omernik *et al.*, 1981; Schlosser and Karr, 1981; Lowrance *et al.*, 1984; Cooper *et al.*, 1987; Osborne and Wiley, 1988; Richards and Host, 1994; Richards *et al.*, 1996; Roth *et al.*, 1996; Johnson *et al.*, 1997; and Wang *et al.*, 1997). In some cases land cover in the watershed was found to be more important than in the riparian corridor in determining turbidity and nutrient concentrations in streams (Omernik *et al.*, 1981; Osborne and Wiley, 1988; Gove and Edwards, 2000). In other studies, riparian characteristics explained more variation in water chemistry parameters and biotic condition than watershed data (Johnson *et al.*, 1997; Lammert and Allan, 1999). The importance of riparian areas as filtering mechanisms for trapping sediment from agricultural fields before reaching a stream (Schlosser and Karr, 1981; Lowrance *et al.*, 1984; Cooper *et al.*, 1987) and in determining stream habitat and biotic characteristics (Karr and Schlosser, 1978) is well documented. The differences in conclusions from these studies indicate that the influences of landscape factors on streams are complex and may be operating at both riparian and watershed scales (Richards *et al.*, 1996; Lammert and Allan, 1999). To further confound results, many studies that have looked at relations between watershed and riparian land cover versus biota have included the land cover in the riparian corridor as part of the watershed land cover, making it difficult to separate the effects of riparian land cover from the watershed land cover. In addition, the land use/cover data used in some of these studies were based on air photo interpretations, from 10 to 20 years ago, with a resulting resolution of 1 to 16 ha and these differences may have contributed to the inability of separating the impacts of watershed from

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riparian land cover. These studies show clearly the need for additional work with higher resolution and more current land-use/cover data using refined methods, as demonstrated by Moser *et al.* (2000).

This study focuses on describing the influences of watershed and riparian corridor land cover, and reach-scale habitat characteristics on biological communities using more recent and higher resolution data for eastern Wisconsin warmwater streams. The objectives of this study were: (1) to examine relations between land-use/cover characteristics at different scales versus stream habitat and biological communities; (2) to compare land use/cover within different areas of the riparian corridor and watershed on stream quality to determine if differences exist between location of land cover and their affect on stream quality; and (3) to identify the importance of continuity and width of an undisturbed riparian corridor to stream quality. These objectives were achieved by using Spearman rank correlation on individual pairs of biological and environmental variables and by applying multivariate analyses to examine relations between the biological measures as a group with individual environmental variables.

METHODS

This study was conducted in the Western Lake Michigan and the Upper Illinois River Basins in eastern Wisconsin. The area is dominated by agricultural land use that exists over loamy to clayey ground moraine with little to no relief. Thirty-eight rural watersheds were selected for study (Figure 1) ranging in size from 9 to 71 km^2, with agricultural land covering 20 to 90 percent of the watershed, and population density ranging from 73 to 883 people per km^2 (U.S. Bureau of Census, 1990). All subwatersheds upstream of the sampling sites were independent, and were chosen to minimize the variation in natural biological communities while maximizing the variation in nonurban land use. As a result, selected watersheds were all warmwater, similar in size (second to third order, with most watershed areas less than 50 km^2), had low to moderate gradients, and were all located in the Southeastern Wisconsin Till Plains ecoregion (Omernik, 1987). Previous studies have shown that sites selected based on these environmental variables result in similar fish assemblages in the absence of human perturbations (Lyons, 1996). A single reach was sampled for each stream and was located near the base of the watershed in order to capture the influences of the full range of watershed and riparian-corridor land use and aquatic habitat on biological communities.

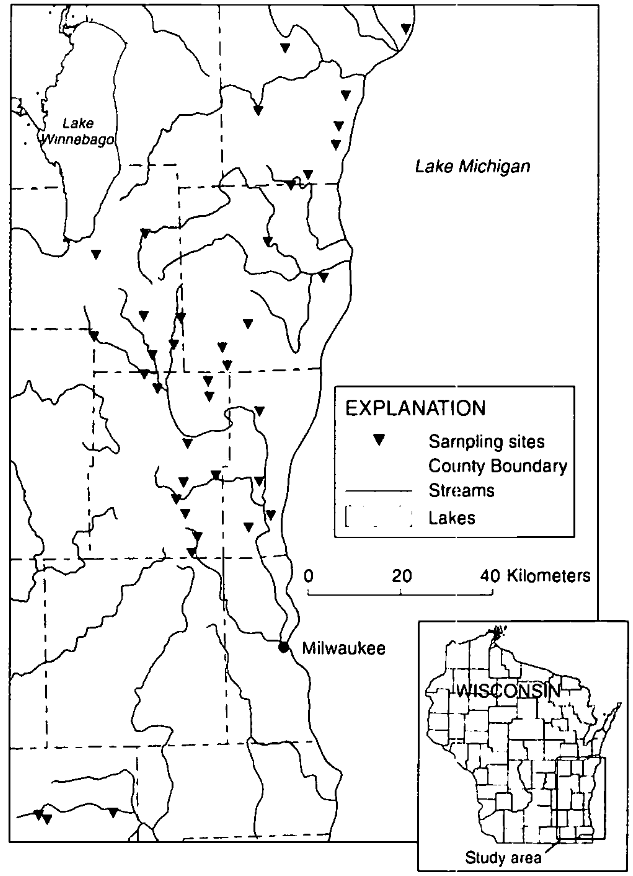

Figure 1. Location of Sampling Sites for 38 Warmwater Agricultural Streams in Eastern Wisconsin.

Biota and Reach-Scale Habitat Data Collection

Fish data were collected at each stream in 1998. Sampling reaches were about 35 times the mean stream width, a length sufficient to characterize the fish assemblage and to encompass about three meander sequences (Lyons, 1992a; Simonson *et al.,* 1994). Sampling reaches ranged from 104 to 242 m in length. Fish sampling occurred between late May and late August, when low stream flows facilitated sampling effectiveness and large-scale seasonal fish movements were unlikely to occur (Lyons and Kanehl, 1993). The entire length of each reach was electrofished with either two backpack units in tandem or a single tow-barge unit with three anodes (Lyons and Kanehl, 1993; Simonson and Lyons, 1995). Efforts were made to collect all fish observed and all captured fish were identified and counted.

Previous studies have shown that this sampling procedure yielded an accurate and precise picture of the fish community (Lyons, 1992a). Several fish metrics were calculated from the community data

(Table 1). A fish diversity index was calculated according to Brewer (1979), and the index of biotic integrity (IBI) for warmwater fishes was calculated for each stream as described in Lyons (1992b).

TABLE 1. Variables Used to Describe Fish and Macroinvertebrate Communities at Sites.

Metric	Range of Values	Median
Percent Tolerant Fish	22.21-95.84	63.96
Percent Intolerant Fish	0-5.48	0
Percent Insectivorous Fish	16.16-96.9	49.46
Number of Fish Species	6-21	12
Fish Density (fish/100 m^2)	27.65-864.9	233.25
Fish Diversity[1]	0.4-2.96	2.05
Index of Biotic Integrity[2]	0-55	30.0
Number of Macroinvertebrates	76-506	148
Number of Macroinvertebrate Species	13-34	18
Percent EPT Individuals[3]	0.76-85.2	35.7
Percent EPT Species	4.6-56.3	28.9
Macroinvertebrate Diversity[4]	1.21-3.0	2.18
Hilsenhoff Biotic Index[5]	3.95-8.11	5.30

[1]Values range from heavy pollution (< 1), light pollution (2 to 3), to no or very slight pollution (> 3) (Brewer, 1979).
[2]Values range from very poor (0 to 19), poor (20 to 29), fair (30 to 49), good (50 to 64), to excellent (65 to 100) (Lyons, 1992b).
[3]The number of Ephemeroptera, Plecoptera, and Trichoptera (EPT) genera were counted. EPT taxa are generally perceived to be pollution sensitive, indicating better water quality with presence (Lenat, 1988).
[4]Values generally range from 1.5 to 3.5 and rarely surpass 4.5, with higher values indicating healthier invertebrate communities (Magurran, 1988).
[5]Values range from excellent (0 to 3.5), very good (3.51 to 4.5), good (4.51 to 5.0), fair (5.01 to 5.75), fairly poor (5.76 to 6.5), poor (6.51 to 7.25), to very poor (7.26 to 10.0) (Hilsenhoff, 1987).

Macroinvertebrates were sampled from riffles during late October 1998 within each reach using a D-frame net. Standard 'Biotic Index" sampling procedures were followed (Hilsenhoff, 1982; 1987). Three samples were collected and combined for identification. For four of the 38 sites that did not have riffles, macroinvertebrates were sampled from snags, which was the best habitat available at that site. Captured macroinvertebrates were preserved in 95 percent ethanol and sent to the University of Wisconsin-Stevens Point for identification, enumeration, and biotic index calculation. The Hilsenhoff Biotic index (HBI) was calculated (Hilsenhoff, 1982) as measure of stream biotic integrity (Table 1). In addition, the percent of Ephemeroptera, Plecoptera, and Trichoptera (EPT) individuals and percent EPT taxa were determined (Lenat, 1988) and the Shannon Wiener diversity index for macroinvertebrates was calculated (Magurran, 1988).

Habitat sampling occurred within a day of fish sampling. At each reach, 28 habitat variables, encompassing channel morphology, bottom substrate, cover for fish, bank conditions, and riparian land use, were measured or visually estimated along 12 transects using standardized procedures described in Simonson *et al.*, (1994) (Table 2). Percent fines were calculated based on the percent of substrate that was less than 2 mm in diameter (sand, silt, or clay). These procedures yielded data with known levels of accuracy and precision, typically ± 5 to 10 percent (Wang *et al.*, 1996). A habitat rating score was calculated based on these data according to Simonson *et al.* (1994).

Land-Cover Characteristics

Land cover characteristics used in this study are summarized in Table 2. Watershed boundaries for sampling sites were delineated on U.S. Geological Survey (USGS) 1:24,000 topographic quadrangles and digitized using a geographic information system (GIS). Ground truth data were collected during the summer of 1998 by driving the entire watershed and recording and photo-documenting land cover at each road crossing of the stream. Additional field notes were taken to further characterize watershed land use, such as location of active barnyards, rural housing developments, and gravel pits, and types of crops and cropping practices. All ground truth and field notes were used to help interpret and verify land use and land cover from digital orthophotos. Land cover at the watershed scale were compiled from the WISCLAND (Wisconsin Initiative for Statewide Cooperation on Landscape Analysis and Data) satellite-derived land cover map for Wisconsin (Lillesand *et al.*, 1998) using the Level I categories of urban, agriculture, forest, grassland, and wetland (forested and nonforested). These data were collected by the Landsat Thematic Mapper and have a 30-meter ground resolution. Population density at the watershed scale was generated using 1990 Census data for minor civil divisions (U.S. Bureau of the Census, 1990).

Stream networks were digitized for each watershed from digital orthophoto quadrangles (DOQs) using ancillary data (1:24,000 State hydrologic layer and

TABLE 2. Summary Statistics for Watershed, Buffer, and Reach-Scale Variables Measured at, or Calculated for Each Site. (Land cover for the buffer is calculated as percent 30-m buffer and for watershed as percent total watershed. Units of measurement for each variable are listed in parenthesis next to each variable.)

Variable	Scale	Range of Values	Median
Agriculture (percent)	Buffer (0-10 m)	.2-13.26	3.61
	Buffer (10-20 m)	0.64-19.2	8.4
	Buffer (20-30 m)	0.99-21.2	10.4
	Watershed (> 30 m buffer)	18.5-88	59.6
Forest (percent)	Buffer (0-10 m)	1.09-11.8	3.76
	Buffer (10-20 m)	1.0-8.9	3.1
	Buffer (20-30 m)	1.0-8.7	2.8
	Watershed (> 30 m buffer)	1.3-33.7	7.1
Grassland (percent)	Buffer (0-10 m)	1.2-21.9	8.9
	Buffer (10-20 m)	1.4-11.7	5.7
	Buffer (20-30 m)	1.9-11.3	5.1
	Watershed (> 30 m buffer)	0.6-32.9	11.7
Urban (percent)	Buffer (0-10 m)	0.09-1.6	0.43
	Buffer (10-20 m)	0.13-2.6	0.64
	Buffer (20-30 m)	0.16-2.7	0.6
	Watershed (> 30 m buffer)	0.03-14.8	1.74
Forested Wetland (percent)	Buffer (0-10 m)	0.95-23.4	7.3
	Buffer (10-20 m)	0.83-19.6	6.7
	Buffer (20-30 m)	0.86-18.7	6.2
	Watershed (> 30 m buffer)	0.4-17.6	4.3
Nonforested Wetland (percent)	Buffer (0-10 m)	0.05-16.2	6.0
	Buffer (10-20 m)	0.01-14.8	5.5
	Buffer (20-30 m)	0.03-13.1	4.7
	Watershed (> 30 m buffer)	1.05-13	5.0
Stream Length With Gaps in Riparian Vegetation (percent)	Buffer (0-10 m)	3.2-68.6	27.8
	Buffer (10-20 m)	1.9-32	13.8
	Buffer (20-30 m)	1.7-12.4	7.2
Stream Length Without Gaps in Riparian Vegetation (percent)	Buffer (> 30 m)	13.8-93.1	51.2
Mean Length of Gap in Riparian Vegetation (meters)	Buffer (0-10 m)	48.8-276.3	141
	Buffer (10-20 m)	20.8-114.1	51
	Buffer (20-30 m)	20.5-64.2	38.5
	Buffer (0-30 m)	97.5-451.9	215.8
Mean Length of Riparian Vegetation Without Gaps (meters)	Buffer (> 30 m)	76.5-666.5	194.4
Erosion (percent)	Reach	1.5-51.7	15.6
Fines (percent)	Reach	12.6-100	65.5
Width/Depth ratio	Reach	4.4-31.6	10.1
Sediment Depth (cm)	Reach	0.3-45.9	5.2
Embeddedness (percent)	Reach	9.2-100	60.6
Fish Cover (percent)	Reach	0-37.3	4.8
Habitat Score[1]	Reach	35-87	54.5

[1]Values range from poor (< 25), fair (25 to 49), good (50 to 74) to excellent (> or = 75) (Simonson *et al.*, 1994).

County soil survey maps) to guide the process. The 1:24000 streams alone were inadequate to represent the stream network due to differences in scale between the 1:24000 streams and the digital orthophotos, and the fact that the 1:24000 streams are a cartographic portrayal of the hydrologic network and may not accurately represent the true location and extent of these hydrologic features (Leopold, 1994). Digitized streams included perennial and intermittent streams, and ditches that were discernable on the DOQs. The stream networks from published County soil surveys also included these hydrologic features and were used as a guide to better interpret the hydrologic network at the DOQ scale. DOQs were produced from 1992 (1:40,000 scale) and 1995 (1:19,200 scale) NAPP (National Aerial Photography Program) photography with a resulting ground resolution of 1 m and 0.6 m, respectively. The high resolution of the digital orthophotography provided the opportunity to investigate the influences of riparian land cover for narrower buffer widths than previous investigations. The satellite-derived land cover was not used for interpreting riparian land cover due to the limitations posed by the 30-m resolution of the data. The suggested minimum mapping unit for the satellite-derived data is five acres, or 25 pixels, making it too coarse for identifying riparian land cover in a 30-m buffer.

Stream corridors were defined for the entire digitized stream network (perennial and intermittent streams and ditches), using a 30-m buffer on each side of the stream. Land cover for the stream corridor scale was digitized and interpreted from DOQs using ancillary data including County soil survey maps, ground truth from field observations, and the WISCLAND land cover map. The land-use/cover categories for the stream corridor were the same as those for the watershed, however, the urban category was comprised of buildings and roads. It was not possible to interpret a wetland category because of the inability to adequately interpret this category using the DOQs. To accomplish this classification, the wetlands from the WISCLAND land cover data were extracted from the 30-m buffer, and were subsequently interpreted into forested or nonforested wetland categories. In some instances, extracted land cover was interpreted to a nonwetland category as a result of scale differences between the DOQs and WISCLAND land cover data. Additional stream buffers of 10- and 20-m were created and vegetation for these buffers clipped from the interpreted 30-m buffer land cover.

Each land cover type within the 10-, 10- to 20-, 20- to 30-, 0- to 20-, and 30-m buffer was expressed as percent land cover for the 30-m buffer; and for the greater than 30-m buffer, was expressed as percent whole watershed, to examine the differences on stream quality between the buffer areas and the watershed area outside the 30-m buffer (Table 2). To better understand the importance of continuity and fragmentation of riparian corridor natural vegetation, the extent of natural vegetation and associated gaps was measured for the 30-m buffer (Figure 2). Areas of the riparian corridor where the natural vegetation (forest and both forested and nonforested wetland) extended less than 30-m wide on both sides of the stream were considered gaps in the riparian vegetation. Areas where the riparian vegetation extended beyond the 30-m buffer on both sides of the stream were areas without gaps in the riparian vegetation. In many cases, the riparian vegetation extended beyond the 30-m buffer on one side of the stream but less than 30-m on the opposite side of the stream. In these cases, the lack of riparian vegetation on one side of the stream took precedence and these stream segments were treated as if they lacked riparian vegetation on both sides of the stream. The reasoning for this was due to the potential deleterious effects on stream quality that could result from the lack of riparian corridor on only one side of the stream. The presence of riparian vegetation on the opposite side of the stream, while not contributing to potential deleterious effects, would neither negate the potential impacts to stream quality. The gap extent was measured for the 10-, 10- to 20-, and 20- to 30-m buffers with the total gap extent for each of these buffer locations, summarized as the 30-m buffer gap extent and expressed as percent of total stream length and average segment length. The extent of the riparian vegetation without gaps was measured for the stream length where riparian vegetation extended beyond the 30-m buffer and was expressed as percent of total stream length and mean segment length. In most cases streams had either very poor riparian corridors, with long continuous gaps or a high degree of fragmentation, or had very well established riparian corridors, with no gaps in the vegetation for most of the stream length.

Statistical Analyses

Descriptive statistics were calculated for all fish, macroinvertebrate, habitat and land use/cover variables (Tables 1 and 2). Correlation analysis was used to identify relations between biological communities with watershed and riparian-corridor land cover and reach-scale habitat characteristics. Data distributions for many characteristics were not normal, thus Spearman rank correlations, which do not require the assumption of normal distributions, were used (Johnson and Wichern, 1992). Significant correlations are

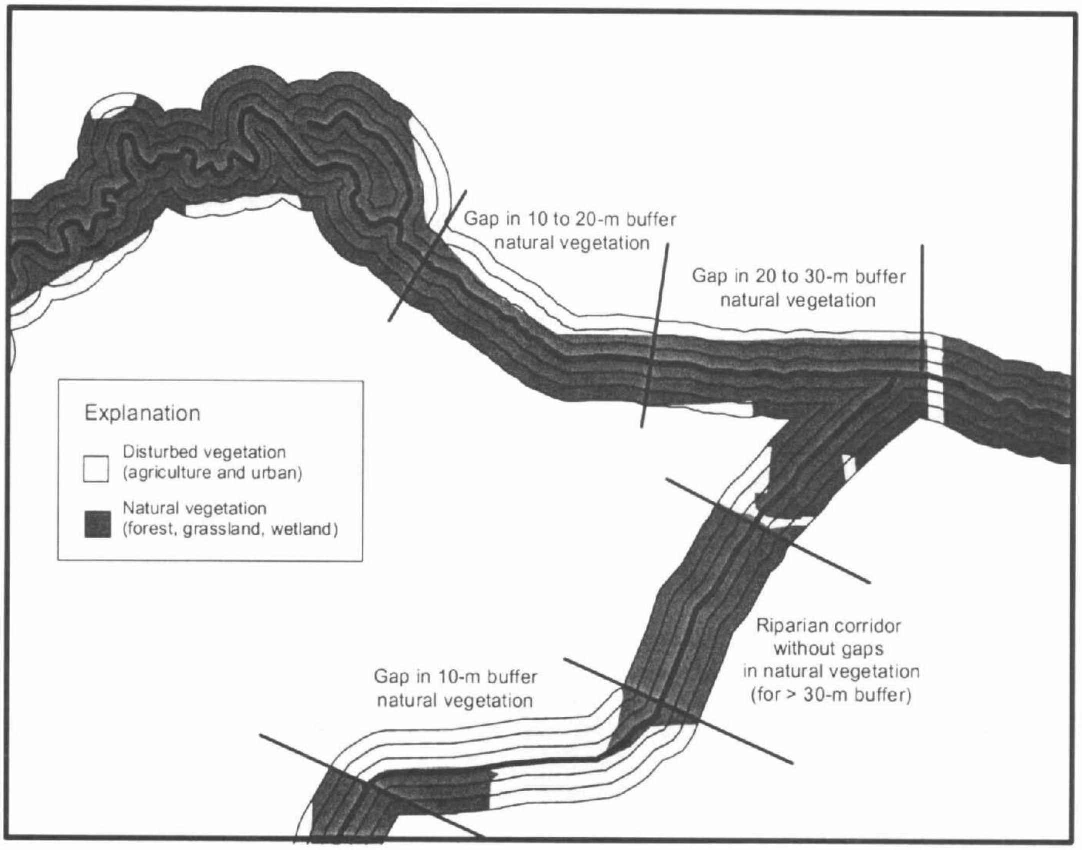

Figure 2. Methods for Characterizing Length of Riparian Corridor Natural Vegetation, With and Without Gaps, for the 10-, 10- to 20-, 20- to 30-, and Greater Than 30-m Buffer.

defined as those where the probability of Type I error is less than 5 percent ($p < 0.05$).

Multivariate statistics, including detrended correspondence analysis (DCA) (Hill, 1979) and canonical correspondence analysis (CCA) (Ter Braak, 1986), were used to further verify relations among biological measures and land use/cover at different spatial scales using the CANOCO 4 program (Ter Braak and Smilauer, 1998). DCA was used to ordinate sites along axes of relative similarity in fish and macroinvertebrate measures without the associated environmental variables. This exploratory tool is used to look at site similarity based on species or species indicators, without associated environmental variables. The scores of the sites in species indicator space (results of DCA axes 1 and 2 based on fish and macroinvertebrate measures) were correlated to each variable for the environmental data set (watershed and riparian-corridor land cover and reach-scale habitat characteristics) to aid in selection of representative variables for CCA. CCA was used to examine relations between watershed, riparian-corridor, and reach-scale habitat characteristics with fish and macroinvertebrate measures. Based on the results of DCA, initial CCA, and correlations among environmental variables, selected environmental variables were deleted from CCA final analyses to avoid inclusion of variables that were highly inter-correlated. Monte Carlo permutation tests were used to determine whether the CCA axes were significant ($p < 0.05$).

RESULTS AND DISCUSSION

The number of fish species collected per site ranged from six to 21 and the number of individuals caught ranged from 28 to 865 per 100 m^2 (Table 1). Fish IBI scores ranged from very poor to good (0 to 55) on a scale of 1 to 100 (Lyons, 1992b). Macroinvertebrate HBI scores, similar to IBI scores, ranged from 8.1 to 4, very poor to very good (Hilsenhoff, 1987) (Table 1). The HBI is based on a 0 to 10 scale, with lower values indicating little to no organic pollution and higher values indicating possible severe organic pollution. The Shannon-Wiener diversity for macroinvertebrates ranged from 1.2 to 3 with higher values indicating healthier macroinvertebrate communities (Magurran, 1988) (Table 1). Habitat scores ranged from 35 to 87, from fair to excellent (Simonson *et al.,* 1994) (Table 2).

Land use/cover in watersheds, outside the 30-m buffer was predominantly agriculture and ranged from 19 to 88 percent (median 60 percent) of the total watershed area (Table 2). Land cover in the 10-m buffer was mainly grassland and ranged from 1.2 to 21.9 percent of the 30-m buffer area (median 8.9 percent), and for the 10- to 20-m, and 20- to 30-m buffer was predominantly agriculture and ranged from 0.6 to 19.2 percent of the 30-m buffer (median 8.4 percent) and 0.99 to 21.2 percent of the 30-m buffer (median 10.4 percent), respectively. Riparian areas may appear to be rather insignificant when considering the size of the area relative to total watershed areas, however research indicates that the influence of the riparian zone on aquatic systems is disproportionate to its total land area (Johnson *et al.,* 1997). The percent of stream length without gaps in riparian vegetation ranged from 13 to 93 percent (median 51 percent). The average length of gaps in the riparian vegetation for the 30-m buffer ranged from 97 to 452 m (median 216 m) while the length of riparian vegetative segments, without gaps, ranged from 76 to 667m (median 194 m) (Table 2). Total stream length ranged from 10 to 84 km (median 25 km).

Correlations varied among aquatic communities, habitat, and watershed and buffer land cover characteristics. Correlations indicate that some factors may be operating at a variety of scales while others may be important at a single scale. Similarly, some factors may affect only one aspect of the biological community where other factors may affect multiple aspects of the community (Table 3). The percent pollution intolerant fish increased with percent forested wetland in the 30-m buffer and with percent forest in the watershed. Similarly, the percent forest in the 30-m buffer was negatively related to percent tolerant fish species. These results indicate that forested riparian corridors and watersheds positively influence intolerant and negatively influence tolerant fish species. The percent insectivorous fish decreased as the percent forest in the 30-m buffer increased. The presence of forested land near the stream, in the 30-m buffer, may have affected benthic productivity of the stream thereby indirectly reducing the percent insectivorous fish. The total number of fish species increased with the percent forest in the 20- to 30-m buffer and watershed, and the percent forested wetland in the 10-m buffer. Fish diversity also increased with the percent forest in the watershed. On the contrary, fish density was negatively related to the percent grassland in the 20- to 30-m buffer and percent urban in the watershed. These measures indicated that higher percentages of forest in the watershed and in the buffer were related to healthy fish communities while near stream grasslands and urban land cover in the watershed had a negative association with the health of fish communities. Relations with the land cover category of grassland may be problematic and require some explanation. The problem is due, in part to the difficulty in separating the use of the land as pasture versus natural vegetation. Although ground truth was collected to help distinguish grasslands from other vegetation types, it was not adequate to distinguish the use for all grasslands encountered in this study, whether they be pasture or left as natural vegetation. However, in this particular study, it is rather common to find near stream grasslands being used for the purpose of pasturing cows.

The HBI scores, higher values indicating poorer water quality, decreased as the percentage of forest in the 30-m buffer and watershed outside the 30-m buffer increased. Similar relations were found for HBI as the percent nonforested wetland in the 10- to 20- and 20- to 30-m buffer, and watershed area, outside the 30-m buffer increased. These results indicated that streams with higher percentages of forested land and non-forested wetland in the watershed, and forested land near the stream (10-m buffer) had less organic or sediment pollution. The HBI scores increased as percent grassland in the 10-m buffer increased, again suggesting that near stream grasslands might have been used for pastures and negatively influenced macroinvertebrate communities. The percent EPT species increased with percent forest in the 30-m buffer and watershed, and with the percent forested wetland in the 10-m buffer. Similarly, the percent EPT species was negatively related to the percent grassland in the 10-m buffer and percent agriculture in the 10- to 20-m buffer. These results suggest that EPT species are positively influenced by forested riparian corridors and watersheds, but are negatively influenced by near stream agriculture and grasslands. The percent EPT individuals also decreased with the percent urban in the 30-m buffer, while the total number of macroinvertebrates collected decreased with percent urban in the watershed. The macroinvertebrate diversity, higher values indicating healthier invertebrate communities, was negatively related to percent urban in the 10- to 20-, and 20- to 30-m buffer and percent urban in the watershed. Urban land cover in the watershed and riparian corridor appears to have a negative impact, not only on EPT species but the entire macroinvertebrate community, as measured by total count. Weigel *et al.* (2000) found macroinvertebrate differences between forested and grassy reaches, but one was not

TABLE 3. Statistically Significant Spearman Rank Correlation Coefficients for Watershed, Buffer, and Reach-Scale Characteristics Versus Fish and Macroinvertebrate Measures (-, not significant; p-value < 0.01 for r > 0.4, p-value < 0.05 for r > 0.3).

Variable	Percent Intolerant Fish	Percent Tolerant Fish	Percent Insectivorous Fish	Fish Species Count	Fish Density	Fish Diversity	IBI	HBI	Percent EPT Species	Percent EPT Individuals	Macro-invertebrate Count	Macro-invertebrate Diversity
Percent Agriculture												
Buffer (10-20 m)	-	-	-	-	-	-	-	-	-0.32	-	-	-
Percent forest												
Buffer (0-10 m)	-	-0.36	-0.51	-	-	-	-	-0.4	0.34	-	-	-
Buffer (10-20 m)	-	-0.36	-0.49	-	-	-	-	-0.4	0.34	-	-	-
Buffer (20-30 m)	-	-0.32	-0.52	0.33	-	-	-	-0.41	0.38	-	-	-
Watershed >30 m	0.34	-	-	0.43	-	0.38	-	0-.46	0.44	-	-	-
Percent Grassland												
Buffer (0-10 m)	-	-	-	-	-	-	-	0.4	-0.37	-	-	-
Buffer (20-30 m)	-	-	-	-	-0.34	-	-	-	-	-	-	-
Percent Urban												
Buffer (0-10 m)	-	-	-	-	-	-	-	-	-	-0.38	-	-
Buffer (10-20 m)	-	-	-	-	-	-	-	-	-	-0.4	-	-0.4
Buffer (20-30 m)	-	-	-	-	-	-	-	-	-	-0.33	-	-0.33
Watershed > 30 m	-	-	-	-	-0.32	-	-	-	-	-	-0.42	-0.37
Percent Wetland – Forested												
Buffer (0-10 m)	0.34	-	-	0.33	-	-	-	-	0.32	-	-	-
Buffer (10-20 m)	0.34	-	-	-	-	-	-	-	-	-	-	-
Buffer (20-30 m)	0.33	-	-	-	-	-	-	-	-	-	-	-
Percent Wetland – Nonforested												
Buffer (10-20 m)	-	-	-	-	-	-	-	-0.32	-	-	-	-
Buffer (20-30 m)	-	-	-	-	-	-	-	-0.34	-	-	-	-
Watershed > 30 m	-	-	-	-	-	-	-	-0.33	-	0.35	-	-
Percent Stream Length With Gaps												
Buffer (0-10 m)	-0.35	-	-	-	-	-	-	0.36	-0.37	-0.34	-	-
Mean Gap Length												
Buffer (10-20 m)	-	-	-	-	-	-	-0.39	-	-	-	-	-
Buffer (20-30 m)	-	-	0.35	-	-	-	-	-	-	-	-	-
No Gap (mean length)	-	-	-	-	0.42	-	-	-	-	-	-	-
Erosion	-	-	-0.32	-	-	-	-	-	-	-	-	-
Percent Fines	-	-	0.33	-	-	-	-	-	-0.32	-	-	-
Sediment Depth	-	-	-	-	-	-	-	-	-0.36	-	-	-
Embeddedness	-	-	0.35	-	-	-	-	-	-0.36	-	-	-

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inherently better than the other and in fact suggested that macroinvertebrates were mostly responding to watershed-scale influences.

An attempt was made to better understand the relations between the continuity and fragmentation of riparian corridor natural vegetation versus fish and macroinvertebrate measures. The percent intolerant fish decreased as the percent of total stream length, with gaps in the riparian corridor natural vegetation, increased for the 10-m buffer. The percent insectivorous fish increased as the average length of gaps in the riparian corridor natural vegetation increased for the 20- to 30-m buffer. Fish density increased with the increase in the average length of riparian vegetation without gaps (> 30-m). The IBI decreased as the gap length increased for the 10- to 20-m buffer riparian vegetation. The HBI increased as the percent of stream length, with gaps in the riparian vegetation increased for the 10-m buffer. Similarly, the percent EPT species and individuals were negatively related to the percent of stream length with gaps in the 10-m buffer natural vegetation. These results indicated that streams dominated by riparian corridors, without gaps and with less fragmentation of natural vegetation, had less organic and sediment pollution, healthier fish and macroinvertebrate communities, and a greater density of fish. There were no significant correlations between watershed population density and any of the biotic factors, therefore this variable was dropped from further analysis.

Correlations reveal important relations between individual pairs of biological and environmental variables, which can be used for identifying environmental variables that are important to biological communities. However, they are not useful for understanding relations among groups of biological and environmental variables. Under natural conditions, relations between biological and environmental variables are complex and rarely just limited to a simple pair relation. In fact, environmental factors are often highly inter-correlated which makes it difficult to draw conclusions from correlation analysis alone. Multivariate analyses can examine relations among multiple biological and environmental variables at the same time and were used in this study to better understand relations among groups of biological and environmental variables.

Multivariate analyses revealed strong relations among biological measures and watershed-, riparian-corridor-, and reach-scale characteristics. Results of DCA indicated that the first two axes explained 73.3 percent of the variation in fish and macroinvertebrate measures with eigenvalues of 0.123 for the first and 0.024 for the second axis. The scores of the sites in species indicator space (results of DCA axes 1 and 2 based on fish and macroinvertebrate measures) were correlated to each variable for the environmental data set (watershed and riparian-corridor land cover and reach-scale habitat characteristics) to aid in selection of representative variables for CCA (Table 4). The average length of riparian corridor without a gap in 30-m buffer natural vegetation was negatively correlated with DCA axis 1 scores, whereas percent grassland in the 10- to 20- and 20- to 30-m buffer were positively correlated with axis 1 scores. The percent grassland in the 10-m buffer, percent agriculture in the 30-m buffer, percent urban in the 30-m buffer, percent stream length with gaps in the riparian vegetation for the 10-m buffer, and percent embeddedness for the reach were negatively correlated with DCA axis 2 scores. The percent forest in all portions of the buffer and watershed, and percent stream length without gaps in the riparian corridor natural vegetation were positively correlated to DCA axis 2 score. These results indicated that these factors were the most important environmental variables influencing the fish and macroinvertebrate communities in this study and were subsequently used in CCA analyses. Many of these factors were also found to have significant relations with biological variables based on the results of correlation analysis.

TABLE 4. Significant Spearman Rank Correlation Coefficients for Detrended Correspondence Analysis (DCA) Site Scores With Environmental Variables (-, not significant; p-value < 0.01 for r > 0.4, p-value < 0.05 for r > 0.3].

Variable	DCA Axis 1	DCA Axis 2
Percent Agriculture		
Buffer (10-20 m)	-	-0.3
Buffer (0-30 m)	-	-0.3
Percent Forest		
Buffer (0-10 m)	-	0.37
Buffer (10-20 m)	-	0.35
Buffer (20-30 m)	-	0.38
Buffer (0-30 m)	-	0.37
Watershed (> 30 m)	-	0.35
Percent Grassland		
Buffer (0-10 m)	-	-0.34
Buffer (10-20 m)	0.3	-
Buffer (20-30 m)	0.36	-
Percent Urban		
Buffer (0-10 m)	-	-0.31
Buffer (10-20 m)	-	-0.34
Buffer (0-30 m)	-	-0.32
Gap – Percent Length (0-10 m buffer)	-	-0.34
No Gap – Percent Length (0-30 m buffer)	-	0.3
No Gap - Mean Length (0-30 m buffer)	-0.3	-
Embeddedness	-	-0.28

CCA procedures, unlike correlation that evaluated relations between individual pairs of biological and environmental variables, examined the relations between groups of biological measures and land cover variables. The eigenvalues for the first four CCA axes were 0.048, 0.015, 0.005, and 0.02, respectively. The first four axes explained 34.8 of the cumulative percentage variance of the fish and macroinvertebrate measures and 97.1 percent of the cumulative percentage variance of the fish and macroinvertebrate measures-environment relation. All variable inflation factors were less than eight and there was no significant covariance among variables used in the analysis. All environmental variables that showed significant correlations with DCA site scores (Table 4) were used in preliminary CCA analyses. However, a number of variables were dropped in subsequent and final runs to avoid inclusion of variables that were highly intercorrelated. Because percent forest in the 10-m buffer was highly correlated with percent forest in the 10- to 20-m buffer (0.96) and in the 20- to 30-m buffer (0.93), the percent forest in the 30-m buffer was selected as the representative variable in CCA and the others were dropped. Similarly, percent urban, percent agriculture, and percent grassland, were each highly correlated with their same cover type within different zones of the 30-m buffer. Therefore, percent land cover for the 30-m buffer was selected as the representative variable in CCA, for these individual cover types. The results of the Monte Carlo test indicated that all axes were statistically significant ($p = 0.015$).

CCA revealed similar results as that of correlation, yet provided additional relations that were not evident from correlation analysis. The most important variables indicated by CCA were, in order, for the first axis, mean length of riparian corridor without gaps in the 30-m buffer, percent grassland in the 30-m buffer, and mean length of riparian corridor for the 30-m buffer with gaps. For the second axis, the most important variables were percent forest in the watershed (> 30-m buffer), followed by percent forest in the 30-m buffer, percent embeddedness, percent urban in the 30-m buffer, mean length of riparian corridor with gaps in the 30-m buffer, percent of stream length with gaps in the 30-m buffer, and percent agriculture in the 30-m buffer. The arrow for an environmental variable points in the direction of maximum change of the environmental variable and its length is proportional to the rate of change for that variable. Environmental variables with long arrows are more strongly correlated with the axes than shorter arrows, so are more closely related to the fish and macroinvertebrate measures (Figure 3) (Ter Braak, 1986). The ordination diagram indicates that forested land cover is positively related to the health of fish communities as measured by number of species, fish diversity, percent intolerant fish, and IBI whether it occurs in the 30-m buffer or in the watershed, outside the 30-m buffer. These fish measures are inversely related to embeddedness, percent agriculture in the 30-m buffer, and the mean gap length in riparian vegetation for the 30-m buffer. Similar relations were found with macroinvertebrate communities, as measured by number of individuals, number of species, invertebrate diversity, and percent EPT individuals and species. These macroinvertebrate measures were positively related to forested land cover in the watershed and negatively related to embeddedness, percent agriculture in the 30-m buffer, and gaps in the 30-m buffer riparian corridor natural vegetation. In other words, near stream agriculture, in the 30-m buffer, has a stronger negative relation to fish and invertebrate communities than agricultural land cover further away from the stream. In addition, riparian corridors with more fragmentation, and longer gaps in the naturally vegetated buffer, were negatively related to health of fish and invertebrate communities whereas riparian corridors with more continuous natural vegetation were positively related to healthy fish and invertebrate communities. The percent urban land cover, and to a lesser degree, percent grassland in the 30-m buffer were positively related to the percent tolerant fish and HBI, and negatively related to the percent intolerant fish, indicating a negative relation to biota. Similar results were found as the percent of total stream length with gaps in the 10-m buffer riparian vegetation increased. When the riparian corridor natural vegetation was more continuous for the 10-m buffer, the density of fish in the stream became higher. Based on DCA and CCA results, near stream agriculture and riparian corridor fragmentation played a stronger role in influencing fish and macroinvertebrate communities than agricultural land cover further away from the stream. Similarly, near stream urban also played a strong role in influencing fish and macroinvertebrate communities as measured by HBI and percent tolerant fish. However, forested land cover played an important role, whether it occurred near the stream or within the watershed.

Results of all analyses indicated that forested land cover plays an important role in influencing fish and macroinvertebrate communities at both the watershed and riparian corridor scales. On the other hand, near-stream grasslands and agriculture were more important in influencing fish and macroinvertebrate communities than those same cover types at the watershed scale. Macroinvertebrate communities were more strongly influenced by urban land cover when located in the 30-m buffer rather than in the watershed, as well as reach-scale characteristics related to sediment deposition. These results clearly show that fish and macroinvertebrate communities

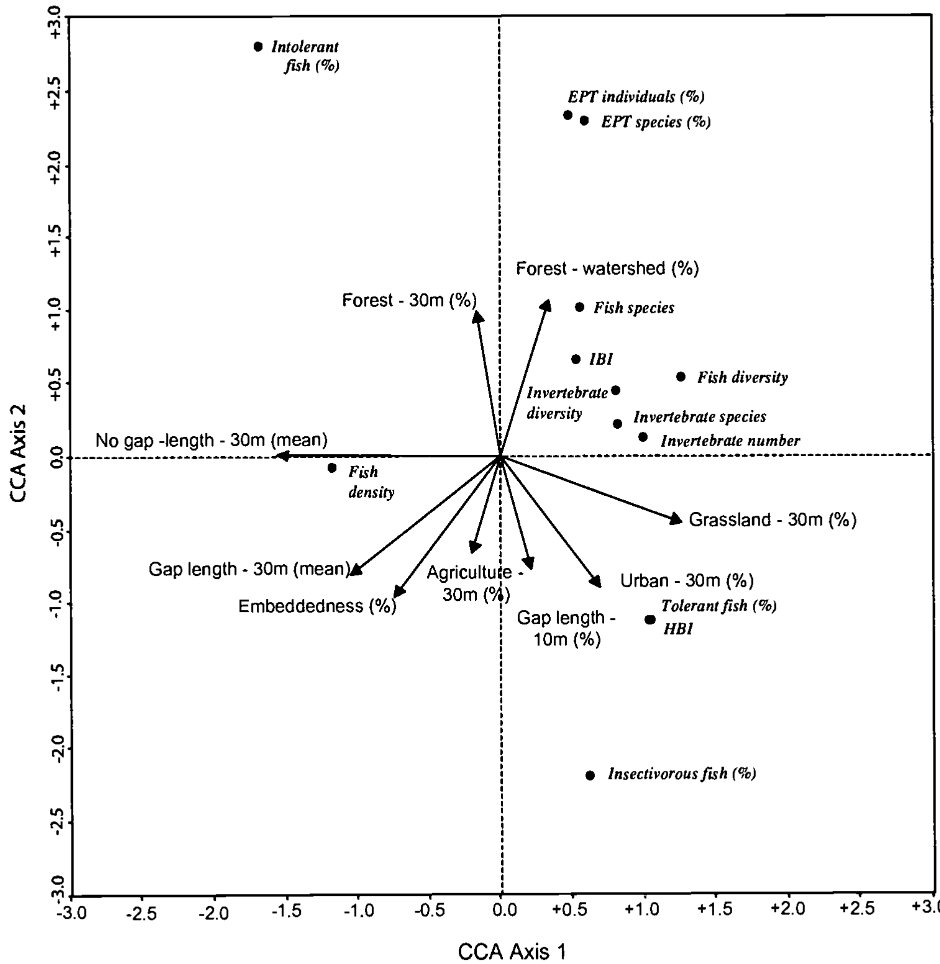

Figure 3. Patterns in Species Indicator/Environmental Variable Relations Shown by Canonical Correspondence Analysis (CCA) Ordination for 38 Warmwater Agricultural Streams in Eastern Wisconsin.

may be influenced by a variety of factors, operating at multiple scales. The continuity and fragmentation of the riparian corridor natural vegetation was also important to the health of both fish and macroinvertebrate communities as indicated by all analyses. Results did not clearly show if differences exist in the location of land cover within different areas of the riparian corridor on stream quality.

CONCLUSIONS

Research has indicated that the influence of environmental factors on stream quality is complex and may be operating at a variety of scales including watershed, riparian-corridor and reach scales (Richards *et al.,* 1996; Lammert and Allan, 1999). Some studies have suggested that the watershed scale is more important than the riparian-corridor scale, while others have concluded that riparian corridors characteristics are more important than watershed factors in predicting stream quality (Omernik *et al.,* 1981; Osborne and Wiley, 1988; Roth *et al.,* 1996; Johnson *et al.,* 1997; Wang *et al.,* 1997; Gove and Edwards, 2000). Differences in results may be due, in part, to differences in resolution and age of land cover data, the scale and extent of the stream network used for riparian-corridor analysis, and whether riparian land cover is summarized as part of watershed land cover, or if watershed and riparian land cover are summarized separately.

In addition, most studies have looked at the amount of land cover that occurred in riparian corridors or watersheds, and have not addressed the continuity or fragmentation of riparian corridor natural vegetation. The type, location, and scale of land cover are not the only factors that influence fish and macroinvertebrate communities. Literature that addressed the suitability of riparian buffers to protect water quality suggested that effectiveness of riparian buffers depends on such factors as width, length, degree of fragmentation, and type, density and structure of vegetation present (Fischer *et al.,* 2000). In this study, the continuity and fragmentation of the riparian corridor natural vegetation were found to play an important role related to fish and macroinvertebrate communities. As the gaps in riparian vegetation increased in length, the health of fish and macroinvertebrate communities decreased, as measured by number of fish species, fish density, fish diversity, IBI, HBI, and EPT species and as indicated by both correlations and multivariate analyses. Results indicated that streams dominated by riparian corridors without gaps and with less fragmentation of natural vegetation have healthier fish and macroinvertebrate communities, and a greater density of fish.

Lammert and Allan (1999) found that land use immediate to the stream predicted biotic condition better than regional land use, but was less important than local habitat variables in explaining the variability observed in fish and macroinvertebrate assemblages. In their study, fish showed a stronger relationship to flow variability and immediate land use, while macroinvertebrates correlated most strongly with dominant substrate. Similarly, Fitzpatrick *et al.* (2000) found fish communities to be more strongly influenced by riparian and to a lesser degree watershed land use and geologic setting. In our study, near stream agriculture was more important in influencing fish and macroinvertebrate communities than agricultural land use in other portions of the watershed. While riparian areas may appear to be rather insignificant when considering the size of the area relative to total watershed areas, research has indicated that the influence of the riparian zone on aquatic systems is disproportionate to its total land area (Johnson *et al.,* 1997). At the reach scale, local habitat measures related to sediment deposition influenced macroinvertebrate and to a lesser degree, fish communities. While these results suggest the importance of agricultural land cover at the riparian scale, forested land cover played an important role at both the watershed and riparian-corridor scales for influencing both fish and macroinvertebrate communities.

The results of this study suggest that stream health, as measured by fish and macroinvertebrate communities, is related to environmental factors at a variety of scales and that resolution of land cover data may need to vary depending on the scale of analysis and the specific question at hand. Although satellite-derived, watershed land cover was important for understanding many land cover/aquatic biota relations, riparian-corridor land cover, derived from DOQs, had the resolution to better understand near stream land cover (within the 30-m buffer) and riparian fragmentation relations with aquatic biota. In order to properly understand these relations, researchers need to use higher resolution and more current land cover data, stream networks defined at a scale that is similar to that of the land cover data, and riparian land cover should not be included as part of watershed land cover if the effects of watershed and riparian land cover versus aquatic biota are to be separated. In addition, the role and importance of continuity and fragmentation of riparian corridor vegetation, as it relates to fish and macroinvertebrate communities is a key factor that should not be overlooked and requires additional study.

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URBAN IMPACTS ON PHYSICAL STREAM CONDITION: EFFECTS OF SPATIAL SCALE, CONNECTIVITY, AND LONGITUDINAL TRENDS[1]

Maeve McBride and Derek B. Booth[2]

ABSTRACT: An assessment of physical conditions in urban streams of the Puget Sound region, coupled with spatially explicit watershed characterizations, demonstrates the importance of spatial scale, drainage network connectivity, and longitudinal downstream trends when considering the effects of urbanization on streams. A rapid stream assessment technique and a multimetric index were used to describe the physical conditions of multiple reaches in four watersheds. Watersheds were characterized using geographic information system (GIS) derived landscape metrics that represent the magnitude of urbanization at three spatial scales and the connectivity of urban land. Physical conditions, as measured by the physical stream conditions index (PSCI), were best explained for the watersheds by two landscape metrics: quantity of intense and grassy urban land in the subwatershed and quantity of intense and grassy urban land within 500 m of the site (R^2 = 0.52, $p < 0.0005$). A multiple regression of PSCI with these metrics and an additional connectivity metric (proximity of a road crossing) provided the best model for the three urban watersheds (R^2 = 0.41, $p < 0.0005$). Analyses of longitudinal trends in PSCI within the three urban watersheds showed that conditions improved when a stream flowed through an intact riparian buffer with forest or wetland vegetation and without road crossings. Results demonstrate that information on spatial scale and patterns of urbanization is essential to understanding and successfully managing urban streams.
(KEY TERMS: urbanization; rivers/streams; geomorphology; land use/land cover; spatial scale; habitat.)

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INTRODUCTION

Urban development, coupled with human population growth, threatens local and global ecosystems (Zipperer *et al.,* 2000). Urbanization of the Puget Sound region has dramatically altered the natural streamflow regime and the physical and geomorphic conditions within stream systems (Booth, 1990; May *et al.,* 1997). As a result of development, once forested land has been replaced with buildings, roads, and lawns. These land cover changes, as well as the extensive changes to the soil profile and the native vegetation community, have altered conditions and processes in lowland streams, which in turn have impaired stream health (Booth, 1991).

The altered physical and geomorphic conditions in urban streams are diverse and complex (Hammer, 1972; Neller, 1988; Booth, 1990; Booth and Jackson, 1997; May *et al.,* 1997; Caraco, 2000; Pizzuto *et al.,* 2000; Hession *et al.,* 2003). In general, urban streams tend to have enlarged cross-sectional dimensions (Hammer, 1972; Caraco, 2000; Pizzuto *et al.,* 2000; Booth and Henshaw, 2001; Hession *et al.,* 2003), accelerated bed and bank erosion (Neller, 1988; Roesner and Bledsoe, 2003), decreased amounts of large woody debris (LWD) and other roughness elements (May *et al.,* 1997; Finkenbine *et al.,* 2000), and simplified morphology (Pizzuto *et al.,* 2000). The grain size distribution commonly shifts to smaller sizes in urban streams (Booth and Jackson, 1997); conversely, smaller grain sizes may be selectively removed in highly urbanized systems where transport capacity greatly exceeds sediment supply (Pizzuto *et al.,* 2000; Finkenbine *et al.,* 2000).

Assessments of the complex physical or biological conditions of urban streams are often attempted by using multimetric indices, measures that integrate

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multiple components to indicate an overall condition (Plafkin *et al.,* 1989; Rankin, 1995; Raven *et al.,* 1998; Barbour *et al.,* 1999; Karr and Chu, 1999). This integrative approach to measuring conditions can help diagnose causes of degradation in complex ecological systems (Karr and Chu, 1999). Another benefit of multimetric indices is their statistical versatility. Because multimetric indices are continuous and can be normally distributed, familiar tests can be applied to identify significant differences in index values (Karr and Chu, 1999).

There is a need for dependable, statistically sound tools to evaluate the amount, location, and distribution of urban land in watersheds. Quantitative methods that link landscape patterns and ecological processes are considered critical to basic ecological research (Turner and Gardner, 1991). Measures of urbanization that go beyond single watershed scale numbers will help to understand and predict the severity and extent of urban effects on stream systems. With better information on the interaction of land cover change and stream ecosystems, it should be possible to improve policies and management strategies for protecting stream integrity in developing areas (Wear *et al.,* 1998). Based on these assumptions, this study had three main objectives: (1) to assess instream physical and geomorphic conditions and their variability within individual urban streams; (2) to measure urbanization using a range of alternative landscape metrics; and (3) to identify relationships between physical stream conditions and various spatial scales and degrees of urbanization.

METHODS

Study Streams

Multiple stream reaches were studied within four watersheds in the Puget Sound Lowland region with similarities in watershed size, surface geology, and relief ratio (Figure 1). In total, 70 sites were sampled: 7 in Juanita Creek, 28 in Swamp Creek, 22 in Little Bear Creek, and 13 in Thorndyke Creek. The watersheds range from approximately 17 to 60 km^2 and are predominantly underlain by glacial till (Table 1). The relief ratios, defined as the difference in elevation between the highest and lowest points of the watershed divided by the length of the watershed (Dunne and Leopold, 1978), range from 11 to 23 m/km.

The study watersheds were selected to span a range of urban land cover (Table 1). Thorndyke Creek, on the western side of the remote Olympic Peninsula (Figure 1), served as a reference stream. Thorndyke Creek's watershed has very little development and is predominantly forested, although some logging has occurred in the watershed. Approximately 20 percent of the upland areas of Thorndyke Creek's watershed were logged at the time of this study. The watershed of Juanita Creek, which flows into the northwest side of Lake Washington, is highly urbanized. Little Bear Creek and Swamp Creek, also tributaries in the Lake Washington watershed system, both have moderate levels of urbanization. Forested areas in all watersheds are predominantly second-growth or third-growth forests.

Field Methods

Physical conditions in the study streams were sampled using a rapid assessment technique during the summer of 2000. The assessments were based on average conditions within 100 m reaches. Assessment reaches were randomly located approximately every 300 to 500 m along the mainstem channel, except where access was prohibited, in wetlands, or in nonalluvial reaches (e.g., reaches constrained by bank armoring). The location of the downstream end of each sample reach was located using a Garmin 12XL global positioning system (GPS) unit. These point locations are hereinafter referred to as sites.

Quantitative and qualitative measures were taken to describe channel morphology, estimate channel dimensions, and characterize bed substrate. Bed morphology was classified as cascade, step-pool, plane bed, pool-riffle, or dune-ripple (Montgomery and Buffington, 1998). The presence of sediment storage bars was recorded (Knighton, 1998). Channel planform was classified as straight, meandering, or braided (Leopold and Wolman, 1957). Gradient was measured at each site using a clinometer and stadia rod. Bankfull width and average bankfull depth were measured at one representative riffle and pool feature for each site, using a tape and stadia rod. An estimate of bankfull cross-sectional area was derived from the product of average bankfull width and depth. An enlargement ratio was then calculated as the ratio of the measured channel size to an expected channel size determined from a regional regression of bankfull cross-sectional area to watershed size for nonurban streams (Booth, 1990). Streambank stability was visually evaluated and ranked as stable, slightly unstable, moderately unstable, or unstable (Henshaw and Booth, 2000). Channel spanning pools with a residual depth greater than one-fourth of the bankfull depth were tallied (Montgomery *et al.,* 1995). Large woody debris pieces were tallied within the active bankfull channel if LWD was at least 25 cm in diameter and 3 m in

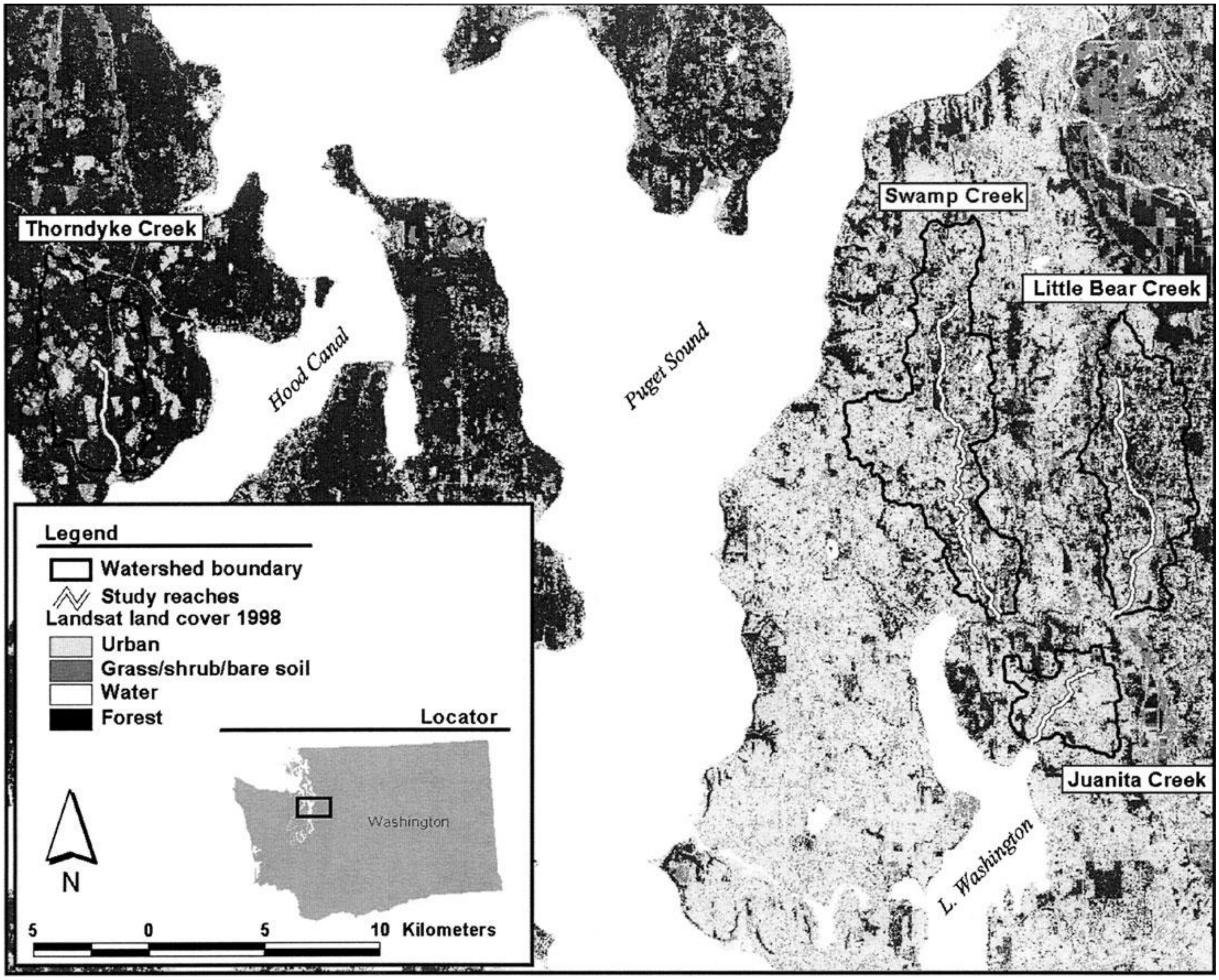

Figure 1. Locator Map With Study Watersheds and 1998 Land Cover (Center for Water and Watershed Studies, 1998) Using a Classification Simplified From Hill *et al.,* 2003.

length (Scholz and Booth, 2001). The structural complexity of the stream was visually assessed and ranked in four classes from excellent/complex to poor/simple. The structural complexity rank was based on the sites' diversity in channel geometry, planform, types of pool and riffle features, and overall structure (McBride, 2001). Substrate size of active riffles or bar features was determined using the pebble count method, where 100 clasts were selected randomly from the riffle or bar surface (Wolman, 1954). Both substrate embeddedness (Barbour *et al.,* 1999) and substrate cementation of riffle features (McBride, 2001) were each ranked in four visual classes (poor, fair, good, and excellent). The substrate embeddedness rank was based on an assessment of the embeddedness of approximately 10 randomly selected individual clasts. All measurements were made by the same observer and under similar base flow conditions between July and September 2000.

Spatial Methods

A GIS based spatial analysis was used to characterize the landscape contributing to each sampled site. Several spatial data sources were employed to characterize the study watersheds, including land cover (30 m, Landsat; Center for Water and Watershed Studies, 1998; Hill *et al.,* 2003); elevation (10 m, 1:24,000 digital elevation model; University Libraries, 1999); wetlands (1:24,000, National

TABLE 1. Watershed Sizes and Land Cover Distributions.

		Land Cover Distribution							Surface Geology			
Stream	Watershed Size (km^2)	Intense Urban (percent)	Grassy Urban (percent)	Forested Urban (percent)	Grass/ Shrub (percent)	Bare Soil (percent)	Forested (percent)	Wetland (percent)	Glacial Till (percent)	Glacial Outwash (percent)	Alluvium (percent)	Other (percent)
Juanita	17.4	9	32	39	6	0	13	2	45	46	0	9
Swamp	58.8	11	27	28	8	1	21	4	79	16	4	1
Little Bear	40.3	5	15	32	7	1	37	2	68	29	3	1
Thorndyke	31.0	2	7*	11*	5	0	72	3	77	20	2	0

*Recently logged areas in Thorndyke Creek appear as grassy urban and forested urban.

Wetlands Inventory; U.S. Fish and Wildlife Service, 1987-1989); and roads (1:24,000; Puget Sound Regional Council, 1997, unpublished, data). The land cover classification is a 30 m grid that distinguished a total of seven categories, three of which were "urban" categories – intense urban land, grassy urban land, and forested urban land. Intense urban lands are areas with the highest amounts of pavement, and total impervious area (TIA) is approximately 92 percent in this category (Hill *et al.,* 2003). Grassy urban lands areas distinguish areas with high amounts of pavement and moderate amounts of grassy or shrub vegetation, and TIA is approximately 74 percent (Hill *et al.,* 2003). Forested urban lands are areas with high percentages of pavement and moderate amounts of forest vegetation, and TIA is approximately 34 percent (Hill *et al.,* 2003).

Three landscape zones were delineated for each sampled site to characterize the magnitude and potential hydraulic connectivity of urban land at different spatial scales. Often, the primary zone of interest is the watershed, the total contributing area of the landscape. Subwatersheds were delineated for each sampled site using GIS. A second delineated zone was the "buffer," which was defined as the total riparian area upstream from the site location (Figure 2). Two buffer zones of different widths, 100 m and 200 m, were created. The third zone of interest was the "local" zone, defined as that portion of the total watershed uphill from the site location and within a specified distance (Figure 2). Two local zones of different sizes, with boundaries 500 m and 1,000 m from the sampling site, were created. Both buffer and local zone boundaries were determined along topographic flow paths. The areas of the buffer and local zones were not extracted from the subwatershed zones, a method preferred by some researchers (Fitzpatrick *et al.,* 2001; Wang and Kanehl, 2003). The methodologies used in this study for buffer and local zones are similar to other spatial analyses (Roth *et al.,* 1996; Allan *et al.,* 1997; Schuft *et al.,* 1999), particularly those of Morley and Karr (2002).

Following the delineation of the three spatial zones (subwatershed, buffer, and local), landscape metrics that characterize both the magnitude of urban development and the connectivity of urban land were defined. Magnitude metrics included the fractions of urban land categories in a given spatial zone (Table 2). Connectivity is broadly defined as "how spatially or functionally continuous a patch, corridor, network or matrix of concern is" (Zipperer *et al.,* 2000, page 687). The connectivity metrics (road density, median flow path length, and upstream distance to road) specifically addressed the hydraulic connectivity of urban land to the channel network within a particular spatial zone, as listed and described in Table 2.

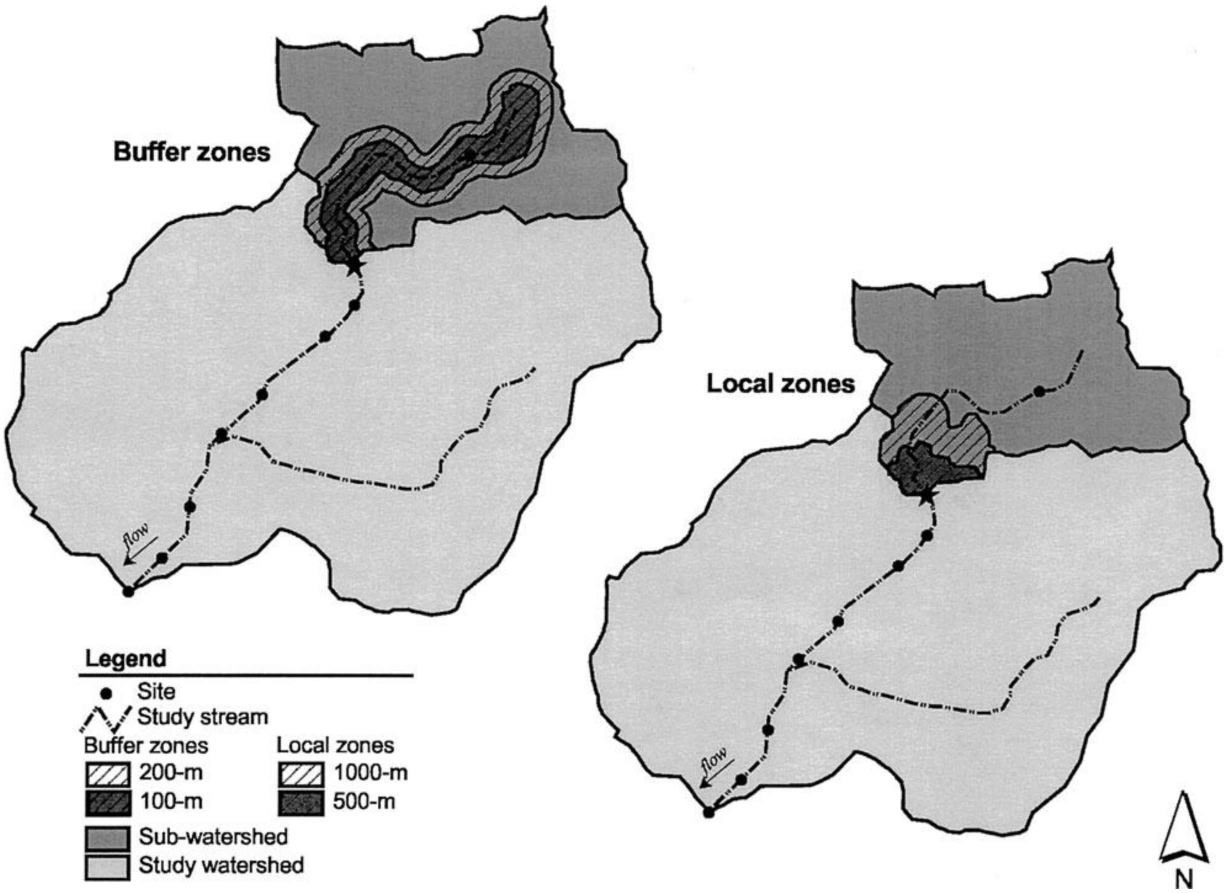

Figure 2. Conceptual Illustration of the Types of Spatial Scales Used in This Study.

TABLE 2. A List of Landscape Metrics, Their Units, and Detailed Descriptions.

Name	Unit	Description
Magnitude Metrics		
Intense Urban Land (IU)	Percent	Proportion of intense urban land
Intense and Grassy Urban Land (IGU)	Percent	Proportion of intense urban land and grassy urban land
Total Urban Land (TU)	Percent	Proportion of all three urban land categories (intense, grassy, forested)
Connectivity Metrics		
Road Density (RDD)	km/km^2	Total road length within a zone divided by the area of the zone
Median Flow Path Length (MFPL)	m	Median value of all flow path distances from each pixel of urban land to the closest stream channel
Upstream Distance to Road (UPRD)	m	Distance between a site and the closest upstream road crossing

The road density metric (RDD) represents the overall connectedness of the landscape regardless of the type of land cover. Roads are typically conduits for stormwater either via pipes or roadside ditches. The median flow path length metric (MFPL) is a measure of the proximity between urban areas and the stream channel network, regardless of the road network. The upstream distance to road metric (UPRD) represents how connected a particular stream site is to the nearest significant road crossing, which usually coincides with a point source of storm water runoff from the road or an adjacent urban area.

The intactness of the riparian buffer between consecutive sites was described via spatial analysis to evaluate longitudinal downstream trends. The intactness of the riparian buffer was defined by two measures: (1) the proportion of forest and wetland areas remaining in the 100 m buffer between any two sites, and (2) the number of road crossings between any two consecutive sites. The number of road crossings was normalized by the distance between the consecutive sites.

Analytical Methods

The physical conditions of the study streams were explored and compared using descriptive statistics, parametric tests, and nonparametric tests. Descriptive statistics such as means and proportions were used to analyze gradient, morphology, planform, bar features, and pool abundance. For ordinal variables the median was used to measure the center of the distribution instead of the mean (Afifi *et al.,* 2004). Analysis of variance (ANOVA) was used to test for differences in LWD abundance among the four study streams (Zar, 1984). The Kruskal-Wallis test, a nonparametric ANOVA, was used to test for differences in the ordinal data, including bank stability, structural complexity, embeddedness, and cementation (Zar, 1984). Dunn's nonparametric multiple comparison test was used following the Kruskal-Wallis test to investigate pairwise differences between the streams (Zar, 1984).

A multimetric index was created to compile the measurements of the physical attributes into a single, lumped score of physical stream condition. Six attributes were chosen to be components of the physical stream conditions index (PSCI). Table 3 lists the attributes, their descriptions, and their scoring criteria. These attributes were selected because they are widely observed to vary systematically through a gradient of human influence and because they include many of the responses to urbanization commonly reported in the literature. Channel size and LWD abundance, the two metrics collected as continuous data, were ranked to match the ordinal metrics in four categories. Channel size enlargement values followed a normal distribution, and therefore the ranks were chosen using the mean and standard deviation

TABLE 3. A List of Metrics of the Physical Stream Conditions Index (PSCI) and Their Scoring Criteria.

Parameter	Description	Scoring 1	Scoring 2	Scoring 3	Scoring 4	Correlation With PSCI[1]
Channel Size	Rank based on enlargement above an expected channel size given the watershed size[2]	> 90 percent larger	50 to 90 percent larger	15 to 50 percent larger	15 percent larger	0.26
LWD Abundance	Rank based on quantity of LWD pieces in the 100 m reach[3]	< 5	5 to 9	10 to 14	> 14	0.73
Bank Stability	Qualitative rank of bank conditions in the 100 m reach[4]	Unstable	Moderately Unstable	Slightly Unstable	Stable	0.70
Structural Complexity	Qualitative rank of stream's structural complexity[5]	Poor	Fair	Good	Excellent	0.80
Embeddedness	Qualitative rank of percentage of embedded substrate[6]	75 to 100 percent	50 to 75 percent	25 to 50 percent	< 25 percent	0.59
Cementation	Qualitative rank of compactness of riffle substrate[7]	Poor	Fair	Good	Excellent	0.68

[1]Spearman's correlation coefficient.
[2]Expected channel sizes calculated using regional regression of nonurban streams (Figure 3; Booth, 1990).
[3]May *et al.,* 1997.
[4]Henshaw and Booth, 2000.
[5]Barbour *et al.,* 1999.
[6]Scholz and Booth, 2001.
[7]McBride, 2001.

values. LWD abundance data did not follow a recognizable distribution. Large woody debris abundance was ranked using equal intervals with the highest rank (> 14) based on the average LWD count for the reference stream, Thorndyke Creek. Lacking any conceptual basis to favor one attribute over another, all attributes were ranked with equal weighting, using a numerical scale of 1 to 4, and their individual scores totaled for the index score. Higher scores indicate better physical quality of the stream.

The PSCI was analyzed via simple and multiple regressions with landscape metrics using an acceptable error rate of 5 percent. The PSCI and all predictor variables were checked for normality via the inspection of normal probability plots. No transformations of the PSCI data or the predictor variables were needed. Correlations between the PSCI and its metrics were identified using Spearman's correlation coefficients, and correlations between the landscape metrics were identified using Pearson's correlation coefficients (Zar, 1984). Longitudinal trends in the PSCI were also explored, particularly in comparison to the intactness of the riparian buffer between two adjacent sites. The change in the PSCI score (Δ PSCI) was calculated as the difference in PSCI score between consecutive sites along the stream longitude. Positive values of Δ PSCI indicate downstream improvement, and negative values of Δ PSCI indicate downstream decline. Changes in PSCI score between consecutive sites can be used to test for local effects because the watershed characteristics are virtually identical for consecutive sites. All statistical tests and analyses were performed using SPSS software for Windows (SPSS Inc., 1999).

RESULTS

Physical Stream Conditions

Geomorphic characteristics at all sites were similar in many respects, including gradient, morphologic classification, planform, bar features, pool abundance, and substrate size (Table 4). Channel gradients ranged from 0.3 percent to 2.5 percent. All sites had pool-riffle or plane bed morphology. Channel planform was either meandering or straight; none of the sampled sites were braided. Most reaches had storage features in the form of point or alternate bars. Most of the reaches had an average of four pools per 100 m. Substrate size distributions were very similar among reaches, and the median grain size (d_{50}) ranged from 16 to 45 mm.

Other conditions varied substantially, including bankfull channel dimensions, LWD abundance, bank stability, structural complexity, embeddedness, and cementation (Table 5). Channel dimensions reflected a characteristic relationship with watershed size – as watershed size increased, the channel's cross-sectional area at bankfull increased. The cross-sectional areas of the sampled sites were plotted against watershed area (Figure 3). Thorndyke Creek's channel sizes were larger than expected given the regional regression of non-urban streams (Booth, 1990), which may be a result of current or former logging activity. Large woody debris abundance was significantly different among the study streams ($p = 0.003$, ANOVA). Ranks of bank stability were significantly different among the study streams ($p < 0.0005$, Kruskal-Wallis), but pairwise comparisons showed that several streams had similar rankings (e.g., Juanita and Swamp Creeks; Table 5). Ranks of structural complexity were

TABLE 4. Geomorphic Characteristics of the Four Study Watersheds From Surveys of Multiple Sites.

Stream	n	Mean Channel Gradient (percent)	Range of Channel Gradients (percent)	Proportion of Sites With Pool-Riffle Morphology (percent)	Proportion of Sites With Bar Features (percent)	Mean Pool Count (No./100 m)	Range of Pool Counts (No./100 m)	Substrate d_{50} (mm)*
Juanita	7	1.1	0.8 to 2.0	100	78	4	3 to 6	22.6
Swamp	28	1.1	0.3 to 2.0	75	84	3	1 to 8	45
Little Bear	22	1.2	0.5 to 2.5	73	58	4	1 to 10	32
Thorndyke	13	1.3	1.0 to 2.0	70	100	4	2 to 6	16

*Substrate size at farthest downstream site.

TABLE 5. Variable Physical Characteristics of Study Streams.

Stream	n	Cross-Sectional Area (m^2)[1]	Expected Cross-Sectional Area (m^2)[2]	Mean LWD Count (No./100 m)	Range of LWD Counts (No./100 m)	Median Ranks[3] Bank Stability	Structural Complexity	Embeddedness	Cementation	Mean PSCI Scores (std.)[4]	Range of PSCI Scores	Maximum ΔPSCI Score[5]
Juanita	7	3.6	1.8	4	1 to 13	2^{A}	2^{D}	3^{EF}	2^{HJ}	12.3 (2.5)	9 to 15.5	3.5
Swamp	28	6.1	3.8	6	0 to 25	2.5^{AB}	2^{D}	3^{E}	2^{H}	14.4 (2.4)	10.5 to 19.5	7.5
Little Bear	22	5.6	3.0	9	1 to 24	3^{BC}	3^{D}	4^{FG}	3^{J}	16.7 (3.5)	12 to 22.5	6.5
Thorndyke	13	4.6	2.5	14	0 to 26	3^{C}	3.5	4^{G}	3	19.4 (1.3)	18 to 22.5	2.5

[1]Cross-sectional area of farthest downstream site.
[2]Expected cross-sectional areas calculated using regional regression of nonurban streams (Figure 3; Booth, 1990).
[3]Capital letters (e.g., A) denote which median ranks are not significantly different by Dunn's nonparametric multiple comparison test ($\alpha = 0.05$; Zar, 1984).
[4]Standard deviation values in parentheses.
[5]Δ PSCI is calculated as the PSCI score of one site minus the PSCI score of its upstream neighbor.

significantly different among the study streams ($p < 0.0005$, Kruskal-Wallis), but the three urban streams (Juanita, Swamp, and Little Bear Creeks) were indistinguishable from each other in pairwise comparisons. Ranks of embeddedness were significantly different among the study streams ($p < 0.0005$, Kruskal-Wallis), but some stream pairs had similar rankings (e.g., Juanita and Little Bear Creeks). Ranks of cementation were significantly different among the study streams ($p < 0.0005$, Kruskal-Wallis), but two stream pairs (Juanita and Swamp Creeks, Juanita and Little Bear Creeks) were indistinguishable from each other.

Correlations Among Landscape Metrics at Different Scales

The quantity of urban land cover in the subwatershed showed very different relationships with the quantity of the urban land in the buffer and local zones. Even though the 100 m buffer zone occupies only 16 percent of the subwatershed zone on average, its land cover was nearly indistinguishable from that of the subwatershed. This strong correlation was demonstrated by a correlation of total urban land in the 100 m buffer zone to total urban land in the subwatershed ($r = 0.99$, $p < 0.0005$, Table 6). Because the quantity of urban land in the 100 m and 200 m buffer zones was so closely correlated with that in the subwatershed zone, the buffer zone metrics were abandoned in the subsequent analysis. In contrast, the percentage of urban land was often considerably different between the local zones and the subwatershed zones. Correlations between the subwatershed zones and the local zones were not significant for the intense urban land metric and the intense and grassy urban land metric, but the total urban land in the subwatershed and local zones were correlated ($r = 0.69$ for 500 m local zone, $r = 0.73$ for 1,000 m local zone).

Connectivity metrics were highly correlated with many of the magnitude metrics (Table 6). Generally, watersheds had no “disconnected” urban land, at least in the way connectivity was quantified in this study; none of the study sites had high quantities of urban land and low measures of connectivity. Road density was strongly correlated with the amount of total urban land in the subwatershed by regression analysis ($r = 0.97$, $p < 0.0005$), and the differences in median flow path lengths between the urban streams was slight, ranging from approximately 300 m to 400 m. In contrast, the third connectivity metric (UPRD) varied considerably, ranging from about 100 m to 1,800 m. The UPRD metric was not significantly correlated with any other landscape metrics (Table 6).

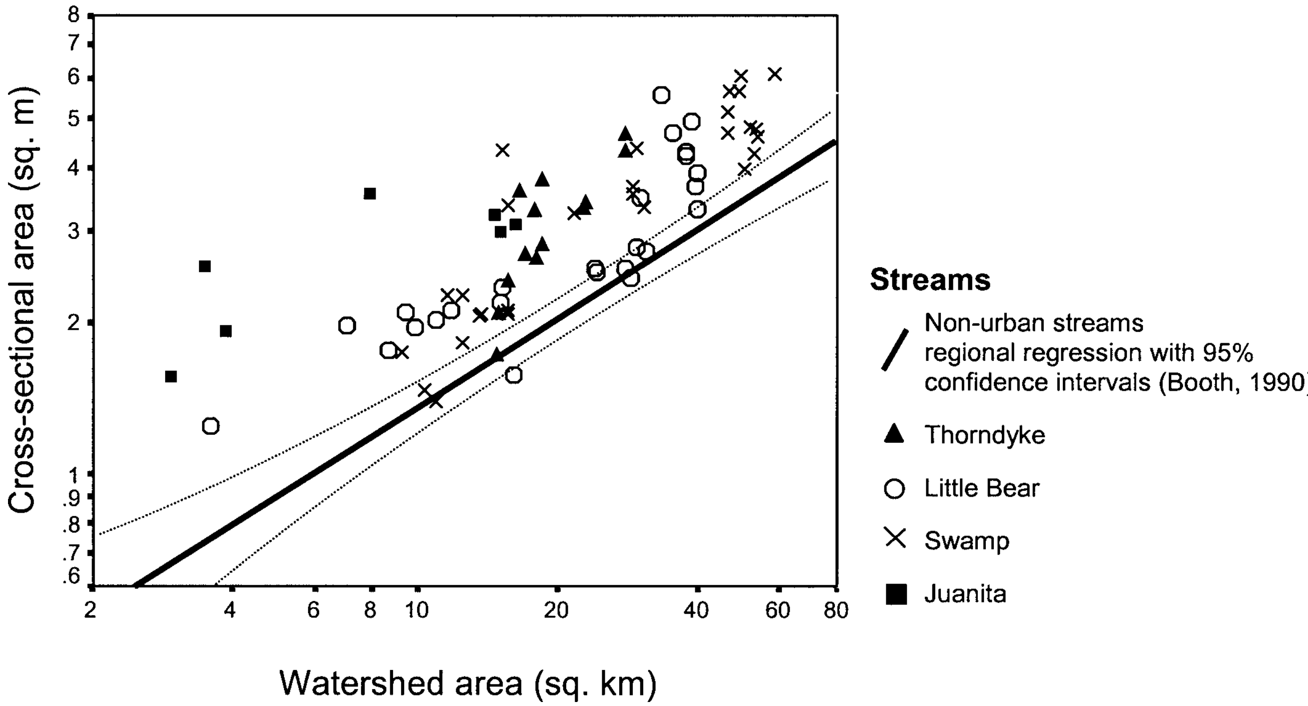

Figure 3. Plot of Channel Cross-Sectional Area Versus Watershed Area for the Study Sites With a Regional Regression Line for Nonurban Streams (Booth, 1990).

PSCI and Urbanization

The mean PSCI scores responded predictably to differences in urbanization. Measured PSCI values ranged from 9 to 22.5 out of a total possible range of 6 to 24. Correlations between PSCI and its metrics indicate that all metrics contributed almost equally to PSCI scores, except for channel size which had a lower correlation coefficient (r = 0.26, Table 3). In general, PSCI scores were greater for watersheds with less urbanization (Table 5). The PSCI showed a significant decline with increasing percent total urban land in the subwatershed zone, though the regression relationship is not compelling (Figure 4a; $R^2 = 0.42$, $p < 0.0005$). When PSCI was regressed with the total urban land within the local zones, the resulting relationships provide some explanation of the variability (Figure 4b and 4c).

Better relationships between the PSCI and the landscape metrics were found using multiple regression techniques instead of single regression models. A better explanation of the variability in the PSCI scores is given by a multiple regression of percent intense and grassy urban land in the subwatershed zone (IGU_{SUB}) and in the 500 m local zone (IGU_{L1}; $R^2 = 0.52$, $p < 0.0005$). Other pairings of urban land magnitude metrics in the subwatershed and local zones provide comparable, statistically significant models. In an attempt to further explain the PSCI, a connectivity metric was added to the regression model. Of all connectivity metrics, only one, upstream distance to a road crossing (UPRD), produced a significant regression model ($R^2 = 0.41$, $p < 0.0005$):

$$PSCI = 20.1 - 11.8\, IGU_{SUB} - 9.4\, IGU_{L1} + 1.7\, UPRD \quad (1)$$

where IGU_{SUB} and IGU_{L1} are in percent and UPRD is in meters.

The sites from Thorndyke Creek were excluded from this regression model because the connectivity metrics (as defined) were not valid in a watershed lacking true urban land cover. For the three urban streams, the regression model in Equation (1) outperforms the regression model with the two magnitude metrics ($R^2 = 0.38$, $p < 0.0005$).

Longitudinal Trends

The PSCI scores were analyzed for longitudinal trends in the three urban watersheds. The variability in PSCI scores among sites in the same urban watershed was high, as compared to the variability in the reference watershed (see measures of standard deviation in Table 5). Swamp and Little Bear Creeks had

TABLE 6. Pearson's Correlation Coefficients Between Landscape Metrics (- not significant, p > 0.05).

Landscape Metric	Abbreviation	IU_{SUB}	IU_{B1}	IU_{B2}	IU_{L1}	IU_{L2}	IGU_{SUB}	IGU_{B1}	IGU_{B2}	IGU_{L1}	IGU_{L2}	TU_{SUB}	TU_{B1}	TU_{B2}	TU_{L1}	TU_{L2}	RDD	MFPL	UPRD
Intense Urban Land (percent)																			
Subwatershed	IU_{SUB}	1																	
100 m buffer	IU_{B1}	0.92	1																
200 m buffer	IU_{B2}	0.95	0.99	1															
500 m local zone	IU_{L1}	-	-	-	1														
1 km local zone	IU_{L2}	-	-	-	0.87	1													
Intense and Grassy Urban Land (percent)																			
Subwatershed	IGU_{SUB}	0.89	0.87	0.87	-	-	1												
100 m buffer	IGU_{B1}	0.83	0.87	0.86	-	-	0.97	1											
200 m buffer	IGU_{B2}	0.84	0.87	0.86	-	-	0.98	1	1										
500 m local zone	IGU_{L1}	-	-	-	0.82	0.73	-	-	-	1									
1 km local zone	IGU_{L2}	-	-	-	0.7	0.8	-	-	-	0.84	1								
Total Urban Land (percent)																			
Subwatershed	TU_{SUB}	0.62	0.65	0.64	-	-	0.87	0.89	0.91	0.37	0.38	1							
100 m buffer	TU_{B1}	0.64	0.66	0.64	-	-	0.88	0.91	0.91	0.36	0.37	0.99	1						
200 m buffer	TU_{B2}	0.6	0.64	0.63	-	-	0.85	0.89	0.9	0.39	0.39	1	0.99	1					
500 m local zone	TU_{L1}	0.21	0.26	0.25	0.47	0.44	0.43	0.44	0.47	0.79	0.68	0.69	0.67	0.7	1				
1 km local zone	TU_{L2}	0.23	0.29	0.27	0.41	0.48	0.46	0.49	0.51	0.71	0.79	0.73	0.72	0.74	0.91	1			
Connectivity Metrics*																			
Road density	RDD	0.4	0.37	0.36	-0.26	-0.32	0.72	0.73	0.76	-	-	0.97	0.93	0.95	0.44	0.54	1		
Median flow path length	MFPL	-0.29	-0.26	-0.28	0.23	-	-	-	-	0.39	0.47	-	-	-	0.34	0.49	0.36	1	
Upstream distance to road	UPRD	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1

*Correlations of connectivity metrics included data from the three urban watersheds only.

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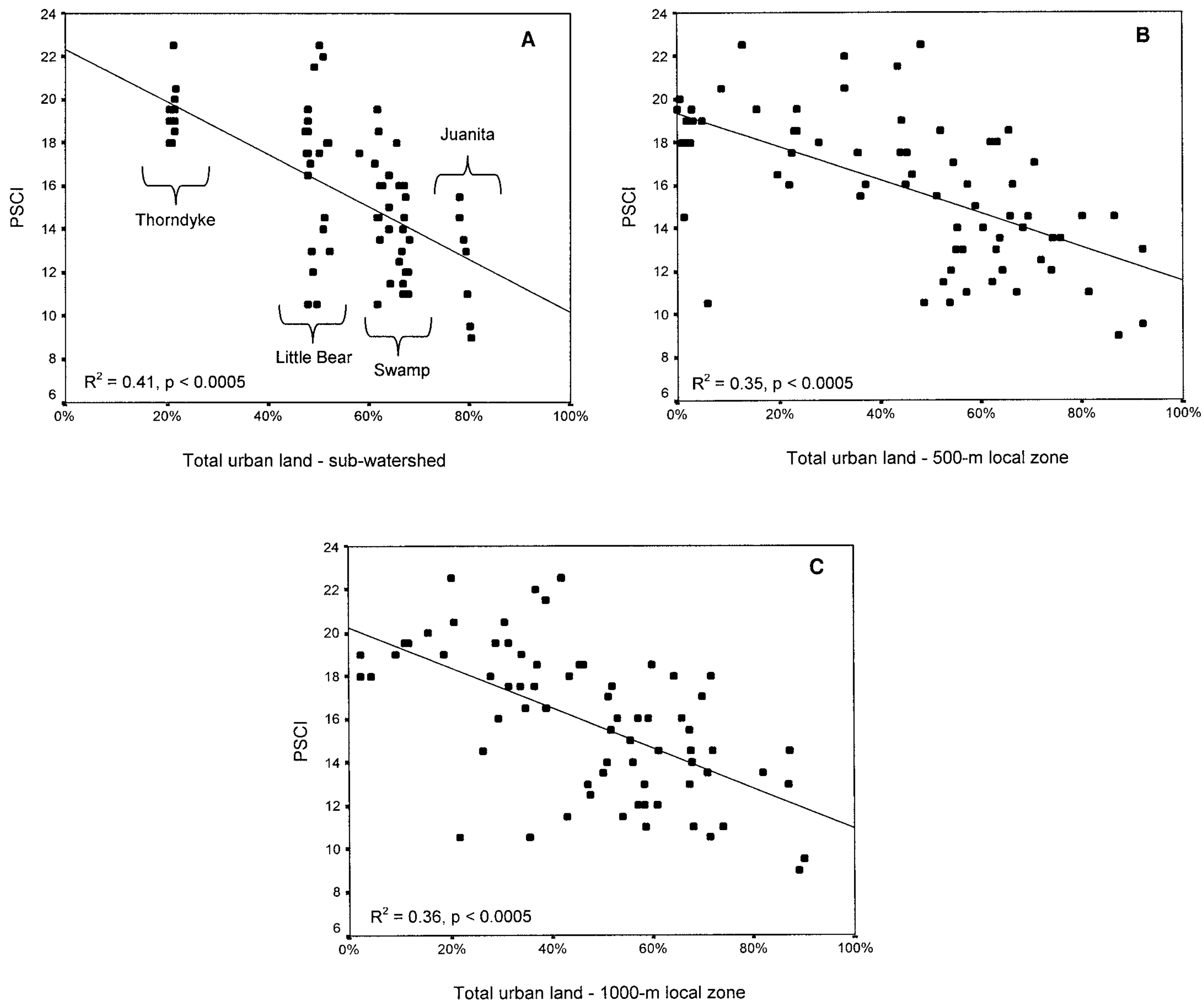

Figure 4. Plots of the Relationships Between PSCI Scores and Urbanization in the (A) Subwatershed, (B) 500 m Local Zone, and (C) 1,000 m Local Zone With Linear Regression Lines.

the greatest overall range in PSCI score (Table 5). Plots of PSCI scores as a function of channel distance demonstrate that conditions changed rapidly between consecutive sites and that continuous downstream trends were not apparent (Figure 5). The change in PSCI scores of consecutive sites were found to range from no change (Δ PSCI = 0) to substantial change (Δ PSCI = 7.5). The change in PSCI scores between consecutive sites was occasionally as great as the total range in PSCI scores within an entire watershed.

The intactness of the riparian buffer explained some of the longitudinal changes in the PSCI score. The PSCI scores were found to significantly improve in the downstream direction (Δ PSCI > 0) when the 100 m buffer between sites was at least 35 percent forested ($p = 0.05$; two-sample t-test with unequal variance) (Zar, 1984). The differences in Δ PSCI were highly significant when sites were grouped using the median value of forest buffer (50 percent), which facilitated a two-sample t-test (unequal variance) with equal sample sizes ($n = 27$, $p = 0.002$, Figure 6a). The presence of road crossings between consecutive sites likely promoted downstream decline in PSCI scores (Δ PSCI < 0; Figure 6b). For consecutive sites with many road crossings between them, the Δ PSCI was often negative. These two riparian factors were not significantly correlated ($r = -0.19$, $p = 0.16$) but appeared to act in concert (Figure 6c). When the buffer between consecutive sites was either not fragmented by a road crossing or fragmented by less than three road crossings per km, the downstream change in PSCI was significantly higher for sites with at least a 50 percent forested buffer (p = 0.08 and p = 0.03, two-sample t-test with unequal variance). When more than three road crossings per km were present, a forested buffer was apparently less effective, resulting in a smaller and less significant relative improvement in PSCI scores (p = 0.10, two-sample t-test with unequal variance).

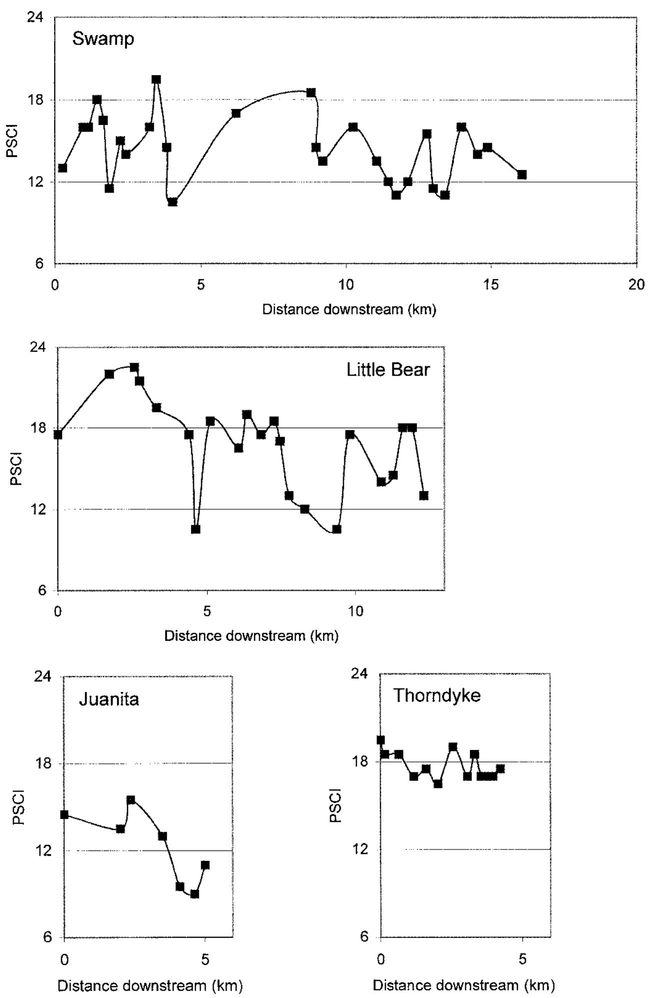

Figure 5. Longitudinal Profiles of PSCI Scores for the Study Streams.

DISCUSSION

Heterogeneity in Physical Stream Conditions

Local instream physical conditions are heterogeneous and are a function of the geomorphic context, the urbanization of the watershed, and the landscape conditions at the local scale. The range of physical stream conditions was greatest for Little Bear Creek and Swamp Creek (Figure 4a, Figure 5, Table 5), suggesting that moderately urbanized watersheds may be more heterogeneous than highly urbanized watersheds or forested, nonurban watersheds. The heterogeneity of moderately urbanized streams is partially explained by the amount of urbanization in the local zone and the intactness of the local riparian buffer. The effects of local urbanization, road crossings, and deforested riparian buffers may be more pronounced in stream systems that have not been overwhelmed by the effects of extensive watershed scale urbanization. Watershed scale urbanization likely sets a maximum attainable best condition, while local and riparian urbanization can further degrade physical conditions. Road crossings appear to be a key point of disruption in urban streams, interrupting the riparian zone and providing a point source for storm water discharges. Other studies have pinpointed roads as key stressors in urban landscapes (May *et al.,* 1997; Marina Alberti, University of Washington, personal communication, December 2003).

Physical Stream Conditions Index

The PSCI effectively integrates a variety of qualitative attributes that are strongly influenced by urbanization into a meaningful, quantitative score. The PSCI functions well as a general measure of the physical integrity in streams, responding in an intuitively reasonable and statistically significant manner to gradients of urbanization. The PSCI correlates well with the proportion of urban land in the subwatershed and local zones (Figure 4). To further evaluate the utility and robustness of the PSCI, however, it needs further validation with other sampling efforts.

The applicability of the PSCI may be limited by the sampling and geographic scope of this study. This index could be used in other Puget Sound Lowland small order (first-order to third-order) streams without hesitation. Applying the PSCI beyond this region or in larger-order streams, however, would not be recommended without first testing its applicability. With that said, most of the PSCI's individual metrics are measures of common symptoms of urban streams in other parts of this country and the world, such as bank instability (Neller, 1988; Trimble, 1997), increased channel size (Pizzuto *et al.,* 2000; Hession *et al.,* 2003), and the loss of LWD (Booth *et al.,* 1997; May *et al.,* 1997).

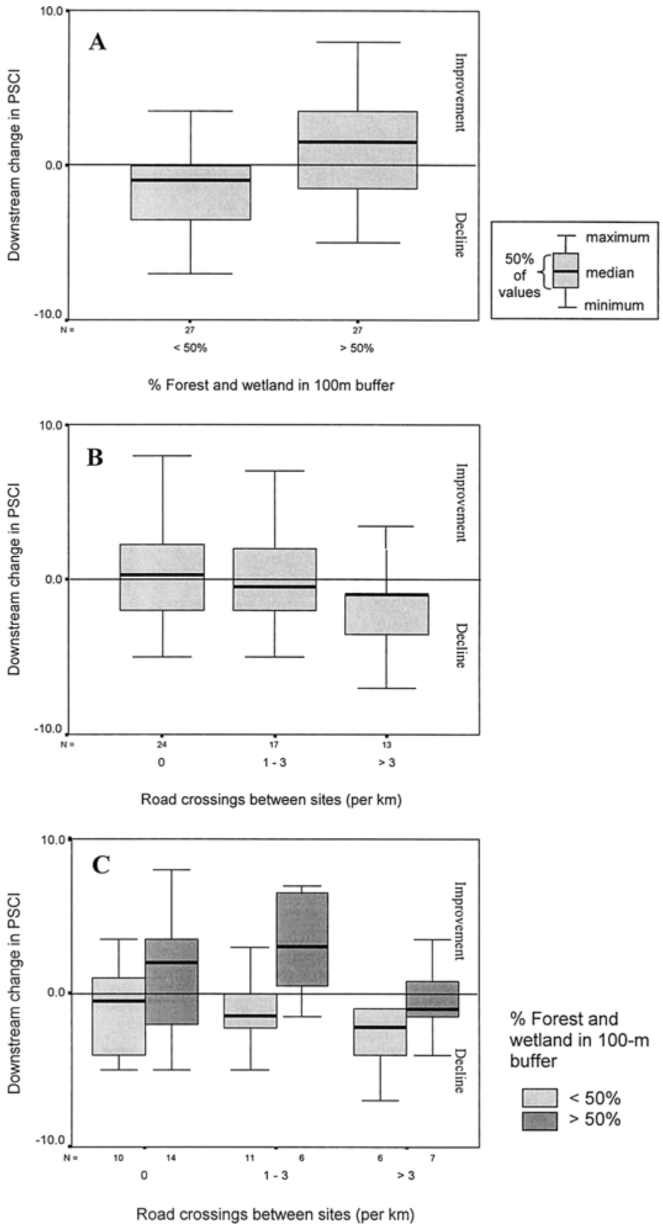

Figure 6. Boxplots of Relationships Between the Change in PSCI Scores (Δ PSCI) and (A) the Amount of Forested or Wetland Buffer Between Consecutive Sites, (B) the Number of Road Crossings Between Consecutive Sites, and (C) the Combined Effect of Buffer Conditions and Road Crossings.

Measuring Urbanization

The quantity, location, and distribution of urbanization can be successfully quantified with relatively simple, GIS based landscape metrics. In some instances, the variety of landscape metrics explored in this study provided a more robust characterization of the urbanized landscape than more commonly used lumped measures of urbanization, such as percent total impervious area. The urbanization of the local zone and the proximity to road crossings provided further explanation of the physical stream conditions of each site; however, some landscape metrics are so closely related that they cannot help decipher stream conditions (i.e., urban land in the buffer zone with urban land in the subwatershed and road density with urban land). Other studies have uncovered similar correlations between land cover in buffers and watersheds (Fitzpatrick *et al.,* 2001; Morley and Karr, 2002; Wang and Kanehl, 2003). Although not useful for better understanding stream conditions, these relationships between landscape metrics provide insight to the nature of the urban landscape.

The pattern of urbanization in the Puget Sound lowlands appears to be fairly homogeneous, as is demonstrated by two key results. First, urban land is evenly distributed throughout the study watersheds in relation to the stream network. The nearly equivalent median flow path lengths found in this study indicate that urban land is not clustered near or far from any particular stream channel, which is consistent with the finding that the urbanization of riparian buffers mirrors the urbanization of the entire watershed. Second, urban areas appear to be equivalently connected to the stream network, as measured by the connectivity metrics in this study. An increase in urban land leads to an increase in the number of roads connecting urban areas to stream channels. The minimal variation in median flow path lengths among the urban streams also demonstrates that urban areas have uniform connectivity.

In contrast, other studies have found variations in connectivity to be an important and influential factor (Bledsoe and Watson, 2000; Wang *et al.,* 2001; Walsh, 2004). Bledsoe and Watson (2000) have studied the change in stream power associated with increased impervious areas and have found it to be sensitive to the connectedness of those impervious areas. A study of Wisconsin urban streams found that the amount of connected impervious area was the best measure of urban impact to several biotic and physical indicators (Wang *et al.,* 2001). A recent study in the Puget Sound region has determined that the number of road crossings per stream kilometer best predicts biological integrity in streams of 42 drainage basins (Marina Alberti, University of Washington, personal communication, December 2003). The importance of connectivity in predicting urban stream conditions may be a function of how connectivity is measured. Road density, as a metric of connectivity and as a coarse estimate of the extent of hydraulic connections to the channel network, did not provide any additional explanatory power in this study. If connectivity can be measured using more detailed information on storm water drainage, as in Wang *et al.* (2001), it may be an important predictor of stream health.

Important Zones of Influence

Results suggest that physical stream conditions are impacted by urbanization in the subwatershed and local zones to nearly equivalent degrees. The regression of PSCI against subwatershed and local zone intense and grassy urban land revealed that these landscape metrics were equally important predictor variables. The combination of intense and grassy urban land (IGU) had a better regression relationship with PSCI than the total urban land (TU) that includes forested urban land. Forested urban lands with low total impervious areas likely do not impact urban streams as severely as the other urban land areas. These results mirror those of other studies. Although watershed conditions are undeniably influential, many studies have identified a disproportionate influence of the local or riparian zone (Steedman, 1988; Wang *et al.,* 2001; Morley and Karr, 2002; Wang and Kanehl, 2003). A similar study of several Puget Sound streams found that biological integrity was equally well predicted by urbanization in the watershed and by urbanization in the local area (Morley and Karr, 2002). Wang *et al.* (2001) found that connected impervious area immediately adjacent to a stream, within either a local zone or a buffer zone, had the strongest influence on an index of biotic integrity and base flow.

In this study, the R^2 values of the various regression models tested suggest that approximately half the variability in physical stream conditions, as measured by PSCI, can be explained by various landscape metrics. Therefore, landscape metrics should not be expected to adequately predict stream conditions, and they cannot be used as a surrogate to instream assessments. Both GIS based analysis and instream assessments of physical or biological conditions are required to evaluate any particular stream system.

Downstream Recovery

Longitudinal trends in the PSCI scores show that partial recovery of physical conditions is possible where a degraded stream flows through an intact forested 100 m riparian buffer. Stream segments with road crossings and without substantial forested riparian buffers tended to have PSCI scores that declined in the downstream direction. The results showed improved physical conditions where the 100 m riparian buffer was at least 35 percent forested. The greatest downstream improvements in physical stream conditions were realized in areas that had few road crossings and substantial forest or wetland riparian buffers within a 100 m corridor of the stream channel.

There are several possible processes acting along a stream channel that could improve physical conditions. Undeveloped riparian zones in the Puget Sound Lowlands typically have active floodplains and riparian wetlands. The roughness of a forested riparian zone and wetland areas can attenuate peak storm flows and reduce specific stream power (Bledsoe and Watson, 2000). If the erosive force of peak flows can be diminished, stream reaches will likely experience less disturbance in their channels, resulting in more stable streambeds and banks. If forested riparian zones and wetlands can significantly slow peak flows and temporarily store storm water, fine sediment suspended or carried in the water column has the potential to filter out and remain deposited in wetlands or on floodplains or within the channel in bars. An intact forested riparian zone also allows the recruitment of LWD and, by definition, precludes many direct anthropogenic impacts, such as channel straightening or streambank armoring.

Management Implications

The results of this study have specific management implications. The amount of development in a watershed is extremely influential on the physical and biological conditions in streams, which necessitates watershed wide land use planning for successful protection of streams. Watershed land use is not the sole determinant of stream conditions, however, and a strategy that imposes only a watershed wide limit on development will be inadequate. Local land cover is extremely important to physical stream conditions, and therefore this zone of the watershed should have high priority in planning and regulations. If urban development can proceed while maintaining intact, undeveloped riparian buffers, the impact of urbanization should be less than from traditional development patterns (Wang *et al.,* 2001). The results also suggest restoration potential for degraded urban streams. If riparian buffers can be reforested and road crossings eliminated or avoided in certain reaches of streams in watersheds with moderate urbanization, partial recovery of a stream's physical integrity is possible.

CONCLUSIONS

This study demonstrates that the effects of urbanization on physical stream conditions are influenced by spatial scale and landscape patterns. Urbanization of both the entire contributing watershed and the part of the watershed closest to the stream appear to have approximately equal weight in influencing a stream's physical conditions, analogous to a prior study of biological conditions in the same region (Morley and Karr, 2002). Urbanization in the watershed is highly influential to streams and likely sets a maximum attainable best condition, yet conditions are strongly modified by the local landscape conditions. Physical conditions can improve downstream from degraded stream reaches if the riparian zone is substantially forested and devoid of road crossings.

Results also highlight the utility of several methodologies used in this study. The PSCI effectively integrates a set of physical attributes, responding in an intuitively reasonable and statistically significant manner to gradients of urbanization in the Puget Sound lowlands. The GIS based analysis generated several landscape metrics that described the quantity, location, and distribution of urban land in the study watersheds and explained much of the variability in physical stream conditions. This study integrated these methodologies to interpret the effects of spatial scale, connectivity, and longitudinal trends in urban streams.

In sum, with better information on the interaction of urbanization and stream ecosystems, policies and management strategies for protecting stream integrity in developing areas can be improved. With more robust knowledge the landscapes can be modified to preserve those streams or stream segments that still function while targeting rehabilitation efforts to those degraded portions of streams that have realistic chances for improvement.

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SOCIAL ACCEPTABILITY OF WATER RESOURCE MANAGEMENT: A CONCEPTUAL APPROACH AND EMPIRICAL FINDINGS FROM PORTLAND, OREGON[1]

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ABSTRACT: Surface water resources in urban areas serve multiple functions ranging from recreation to wildlife habitat. As a result, diverse values influence people's views about resource protection, potentially leading to conflicting interests. In metropolitan Portland, Oregon, natural resource planning has recently focused on habitat restoration as well as stormwater and pollution mitigation, especially through the protection of riparian areas. Due to opposition over proposed regulations in the study region, this research examines public attitudes about an array of resource management efforts. The primary research question is: what is the extent of positive–negative attitudes about water resource protection, and what theoretical dimensions underlie diverse judgments? After empirical survey results are presented, I outline a conceptual approach for future assessments of environmental attitudes while highlighting important value-based dimensions of judgments. Although flexible, the framework allows broad comparisons to advance knowledge about the social acceptability of varied water resource management approaches across diverse places and contexts.

(KEY TERMS: resource geography; environmental attitudes; water management and planning; Portland, Oregon.)

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INTRODUCTION

Streams, lakes, and wetlands provide vital functions for wildlife and people worldwide, however, degradation has greatly diminished their value. In the United States, an estimated 50% of wetlands have been lost, nearly 40% of streams are too polluted for swimming, and 30% of freshwater fish are in danger of extinction (American Rivers, 2009). Riparian areas, or vegetated areas around waterways, are especially critical for ecosystem services such as habitat protection, pollution prevention, and flood mitigation. Yet land development and human activities threaten the ecological functions and social values they offer throughout urban and rural watersheds.

Scientists have long valued river systems for their ecological benefits, while residents tend to value streams for their "enduring aesthetic appeal" (Nassauer *et al.*, 2001, p. 1439). Considering wetlands, which have traditionally been regarded with disdain, Nassauer and colleagues have advocated careful planning for the "cultural sustainability" of water resources. In particular, they stress understanding

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the "ecologically beneficial practices that elicit sustained human attention over time" (p. 1440). Indeed, public perspectives and policy preferences are central to developing socially acceptable approaches to resource protection and management. Viewing public attitudes as an important element of the social pillar of sustainability, this research examines attitudes toward water resource planning initiatives in metropolitan Portland, Oregon, where opposition to proposed regulatory buffers around riparian areas has recently derailed resource protection efforts (Brinckman, 2002; Larson and Santelmann, 2007).

Given the multiple values, actors, and governance approaches associated with resource management, diverging interests and perspectives concerning water and other natural resources often result in culturally-based environmental conflicts (Ozawa, 1996; Berry, 2000). While substantial public support exists for environmental protection generally and for broad objectives such as water quality improvements (House, 1999; Dutcher *et al.*, 2004), attitudinal opposition is common toward specific efforts such as government regulations that threaten local control (Roberts and Emel, 1992; Somma, 1997; Raedeke *et al.*, 2001). Especially in rural areas, politically contentious regulatory approaches have faced strong landowner opposition. Since much of the previous research on environmental attitudes has been conducted with farmers and rural residents, research is needed to understand the extent and array of policy support and opposition among varied residential populations in metropolitan areas.

Despite the fact that three-fourths of the American population lives in cities, far greater attention has been directed toward understanding and protecting aquatic systems in agricultural and rural regions (Kusler, 1988). Research and policy initiatives in urban areas has increased in recent decades as resource values have become apparent, and as water quality and endangered species policies have mandated attention to nonpoint source pollution, riparian buffers, and habitat degradation in cities. Focus on metropolitan regions is a matter of fairness, suggests one rural activist in Oregon, because "Urban landowners are getting a taste of what rural landowners have been facing for years" (Brinckman, 2002: B5). Ultimately, watershed-wide approaches are essential considering the hydrologic linkages between rural and urban communities, wherein human activities in one area can impact those further downhill or downstream.

Employing a case study watershed approach, I examine residents' attitudinal preferences for management alternatives in this paper, specifically focusing on the underlying dimensions of judgments about a variety of goals, entities, and strategies aimed at water resource protection. Building on survey findings and multidisciplinary theoretical perspectives, I then present a conceptual framework to facilitate comparable assessments of attitudes in diverse settings. The primary contributions are two-fold. First, the empirical assessment advances knowledge about the nature and structure of attitudes about water resource management activities, thereby addressing calls by prominent scholars for research on multifaceted environmental views (Dunlap *et al.*, 2000; Stern, 2000; Heidmets and Raudsepp, 2001; Routhe *et al.*, 2005). Second, the conceptual approach facilitates much-needed comparisons across unique places and populations (Tunstall *et al.*, 1999), while maintaining the flexibility to examine human ecological perspectives in specific contexts. As a whole, this paper advances research through improved understanding of public perspectives on a range of choices in water resource governance, and accordingly, their social acceptability and cultural sustainability.

THE THEORETICAL APPROACH

For the purposes of this research, attitudes are defined simply as evaluative positive or negative judgments about an "object" or phenomenon (Thurstone, 1928; Heidmets and Raudsepp, 2001). As public perspectives on assorted objectives, actors, and strategies, environmental attitudes are complex and multidimensional. The relationship among attitudes is many-to-many, such that judgments about a single phenomenon may relate to those about multiple other phenomena (Gilbert *et al.*, 1998). Moreover, judgments toward particular phenomena may diverge due to conflicting values and interests.

Environmental attitudes have been investigated at varying levels of specificity, in relation to different targets, actions, and contexts (Whitaker *et al.*, 2006). General dispositions are important because they influence specific views and actions (Heberlein and Black, 1976; Hobson, 2006). In this research, general attitudes are conceptualized as the expressed importance (or not) of a variety of resource management objectives, which reflect underlying values and value orientations. As patterns of basic beliefs about what is important, value orientations mediate the relationship between core values and attitudes (Whitaker *et al.*, 2006).

At the most abstract level, values define what is important to people broadly and therefore influence judgments about the environment and other matters (Stern and Dietz, 1994; Stern, 2000; Schaaf *et al.*, 2006). Schwartz (1994) describes two basic value groups: traditional–open-to-change and self-enhancing–

transcending. The first dimension affects particular attitudinal judgments by determining the relative importance attached to maintaining traditions, customs, or the status quo. Both traditional and individualistic values are at the core of conservative political ideologies, such as those underscoring private property rights and a free-market economy. Additionally, selfish–altruistic values affect environmental attitudes by determining the importance placed on the impacts and benefits incurred, whether personally or by society and nature more broadly, under particular resource management regimes.

As a set of basic beliefs pertaining to nature and human capabilities, the "Dominant Social Paradigm" reflects value orientations stressing unlimited resources, continued growth, technologic optimism, and human ingenuity (Pirages and Ehrlich, 1974; Kilbourne and Polonsky, 2005). In contrast, the increasingly prevalent New Ecological Paradigm (NEP) emphasizes the finite nature of resources, limits to growth, and the moral rights of wildlife (Dunlap *et al.*, 2000). These perspectives encompass anthropocentric–biocentric orientations, which are known to influence environmental attitudes, for example, concerning management goals such as public use and enjoyment *vs*. wildlife and habitat protection (Whitaker *et al.*, 2006). Considering human-centered views *vs*. nature-centered views, Whitaker *et al.* have shown in previous studies that utilitarian (use)–preservation (protection) orientations significantly affect judgments about wildlife management strategies. Still other prominent scholars have illustrated how altruistic values combine with biocentric orientations to influence ecological concern (Stern and Dietz, 1994) as well as behavior (Stern, 2000).

Value-based beliefs about nature coalesce with those about society and governance to form judgments about appropriate environmental management and policy approaches (Schwarz and Thompson, 1990; Ellis and Thompson, 1997). Associated with four views of nature, cultural theory outlines two axes of social-political beliefs: orientations toward collective *vs*. individual action (group axis) and top-down prescribed rules *vs.* bottom-up voluntary efforts (grid axis) (Schwarz and Thompson, 1990). Depending on particular cultural rationalities or orientations, people prefer different policy approaches, from government restrictions or market mechanisms to community-based efforts or personal actions. Therefore, in this study I examined attitudes in relation to various entities and management strategies, such as land use regulations and restoration projects as well as incentives and information-based programs. Special attention is placed on government restrictions due to recent opposition in the study area (Brinckman, 2002), in addition to discrepancies between general and specific judgments about regulations.

A previous study conducted in Texas found that general attitudes toward regulating water use were negative due to ideological attachment to private property rights and local control (Roberts and Emel, 1992). However, attitudes toward particular restrictions (specifically, well-spacing requirements for ground-water conservation) were relatively positive because of the minimal impacts associated with their application. Divergences between general and specific attitudes have been reported elsewhere, as in the Chicago area where residents exhibited substantial support for ecological restoration while opposing specific techniques such as the removal of beloved trees (Bright *et al.*, 2002). Thus, value-based ideological sentiments may strongly influence general attitudes while particular personal interests may dominate more specific judgments.

Carman (1998) described the hierarchical nature of attitudes, suggesting that environmental policy support is a function of judgments about resource conditions (such as air or water quality), regulation (such as managed resource use and pollution reduction), and economic issues (such as restricting businesses and environmental spending). Although valuable, Carman's work likens regulation to government protection of natural resources without distinguishing between mandatory regulations and voluntary efforts, economic and market-based measures, or other types of government and non-governmental programs.

Building on previous research and the above-described body of theory, the study presented herein focuses on residents' attitudes about water resource management, including judgments about biocentric and anthropocentric goals, government and nongovernment entities, as well as regulatory and voluntary policy approaches to protecting surface water resources. Economic attitudes about the funding mechanisms for resource protection were also examined, although my approach did not involve the contingent valuation methods employed in previous work (such as Cooper and Keim, 1996; Loomis *et al.*, 2000). Overall, I conceive of individual judgments combining with others to represent distinct aspects (or sub-dimensions) of attitudes, which in turn reflect larger order attitudes about an overarching phenomena (e.g., protecting streams, lakes, and wetlands). The dimensions along which attitudes are expected to vary include individualistic–collective values and biocentric–anthropocentric orientations as well as social-political beliefs about prescribed regulations, economic-based approaches, and voluntary management efforts. Based on these value-based dimensions, the conceptual approach outlined following the empirical results is intended to facilitate broadly comparable studies to advance generalizable knowledge about multifaceted attitudes while addressing the need to consider human ecological perspectives in specific contexts (Barr and Glig, 2005; Flint and Luloff, 2007).

RESEARCH DESIGN

The empirical research entailed preliminary interviews with planning professionals and a self-administered questionnaire of residents. The interviews facilitated the identification of policy-relevant attitude objects toward which judgments were examined. Analysis of survey data then addressed the primary question by examining the nature, extent, and dimensions of attitudes about water resource protection. Before detailing the data collection and analysis methods, the study area is briefly described.

The Study Watershed

The Johnson Creek watershed in Oregon expands urban to rural land uses from central Portland, just southeast of downtown, to unincorporated areas farther east toward Mt. Hood. While low-lying sections of downstream Portland exhibit dense development, degraded riparian zones, and flooding problems, urban growth is expected in suburban communities and the relatively rural headwater areas of the watershed. In the study area and beyond, government and other entities have recently pursued stormwater management, water quality improvements, and aquatic habitat restoration, partly following from federal and state policies that address flood mitigation, nonpoint source pollution, and endangered species. Previous policy initiatives to mitigate flooding and improve water quality resulted in riparian buffer restrictions along waterways (Metro, 1998; City of Portland, 2001). More recently, water resource planning in the region has focused on fisheries and wildlife management due to the listing of salmon species as threatened in the 1990s. Renewed attention to habitat protection has lead to proposals for expanded zones of land use regulation in riparian areas, which have been vociferously opposed by local property rights advocates (Brinckman, 2002; Larson and Santelmann, 2007).

Survey Data and Statistical Analysis

Based on interviews and the scholarly literature, over thirty individual attitudinal judgments were examined in relation to "protecting water resources such streams, rivers, lakes and wetlands" in the greater Portland area (see Appendix for verbatim survey questions and Table 1 for descriptive statistics). Following the Dillman (2000) survey design method, a written questionnaire was implemented with three mailings in 2004. The quantitative attitude variables are ordinal data, with several judgments ranging from "strongly support" to "strongly oppose." Most of the attitudinal variables were measured on six-point response scales, with anchors spanning "very important" to "not important" for the survey questions concerning resource protection goals. The exception is attitudes toward funding mechanisms, which entailed

TABLE 1. Attitudes About Water Resources Protection: Ranked From Most to Least Positive.

Individual Attitudinal Judgments	Negative Attitudes (%)[1]	Mean	St. Dev.	*N*
Drinking water quality	2	1.22	0.70	807
Clean streams, lakes, wetlands*	3	1.61	0.88	807
Protection in general*	3	1.47	0.80	771
Fines land use violations	5	1.34^	0.57	751
Regulations on industrial areas	6	1.64	1.04	748
Education and outreach	6	1.76	1.01	756
Regulations on commercial areas	7	1.81	1.07	750
Fish and wildlife habitat*	7	1.83	1.08	807
Regulations on public parks/spaces	8	1.84	1.11	751
Restricting how develop near water	8	1.73	1.12	764
Restoration projects	8	1.87	1.10	758
Regulations on agricultural areas	10	1.93	1.21	749
Taxes on polluting products	10	1.42^	0.66	738
Restricting tree removal near water	12	1.84	1.32	767
Flood management*	13	2.21	1.20	794
Regulations on residential areas	13	2.20	1.36	752
Restricting plants near water	14	2.02	1.36	740
Fees on new development	14	1.62^	0.72	718
Restricting construction near water	15	2.06	1.38	754
State government	15	2.29	1.24	726
Public use and enjoyment*	15	2.39	1.19	804
Local government	15	2.22	1.31	729
Purchasing land from willing sellers	18	2.28	1.34	732
Financial incentives	21	2.53	1.45	689
Regulations (in general)	21	2.42	1.48	746
Nonprofit organizations	21	2.41	1.47	703
Regional government (Metro)	23	2.54	1.56	728
Voter-approved bond	29	2.05^	0.72	676
Federal government (U.S.)	30	2.71	1.49	722
Personal willingness to pay	31	2.14^	0.96	709
Charges on water/sewer bills	31	2.20^	0.61	722
Businesses (for profit)	45	3.26	1.73	675
Property taxes	49	2.42^	0.62	732
Income taxes	53	2.46^	0.63	729

[1]Negative attitudes represent "opposition" except for items with an asterisk(*), which were expressed as "not important." Percents include multiple responses on negative (e.g., oppose, not important) portion of ordinal scales due to the shorter scales for ^economic measures for which the maximum was 3 instead of 6. See the Appendix for verbatim survey measures.

three-point labels to gauge opposition relative to support for current *vs*. higher levels. Finally, an open-ended question asked survey respondents to further explain their views with written comments about water resource protection and management efforts.

Random samples were drawn from tax assessor databases (to survey resident homeowners) and organizational contact lists (to target participants of watershed and neighborhood organizations). The inclusion of participants in watershed and neighborhood groups reflects additional research questions reported elsewhere (Larson, 2005; Larson and Lach, 2008). All respondents were retained for the analysis due to similar patterns in attitudes across the group participant and nonparticipant samples in terms of their ranked importance and attitudinal dimensions.

The survey response rate was 44% (n = 816), and overall, the respondents were similar demographically to the study area population based on 2000 Census data (Table 2). The sample is mostly White/Anglo due to regional demographics and focus on resident landowners in this study. Given the population targeted and the sample characteristics, caution must be used in generalizing the results to other populations. I therefore emphasize the significance of the findings in relation to theoretical understanding of multidimensional attitudes.

Exploratory factor analysis identified the empirical dimensions of residents' judgments toward water resource protection, as is common in attitudinal research (Bright *et al.*, 2002; Schaaf *et al.*, 2006). Principal axis factor analysis was conducted with an oblique rotation (Direct Oblimin), since dimensions of attitudes about resource protection are likely to be correlated (Carman, 1998). Eigenvalues greater than one indicate significant factors (Kim and Mueller, 1978) worthy of consideration as dimensions of attitudes. Loadings greater than the standard 0.35 are presented, with emphasis on substantial loadings higher than 0.6 (Spector, 1992). The reporting of results centers on the pattern matrix, which produces very clear dimensions for interpretation (Kline, 1994). The primary dimension and sub-dimensions of the factor matrix are mentioned where informative for understanding the nature and structure of attitudes.

TABLE 2. Demographics for the Survey Sample and Study Population.

		Study Area Population[1]	
Demographic Variable	**Survey Sample**	**Multnomah County**	**Clackamas County**
Gender: female	50.3%	50.6%	50.6%
Age (mean years)	53.7	52.9	55.5
Education: Bachelor's degree	26.3%	30.7%	28.4%
Household income (mean)	~US$50,000[2]	US$52,080	US$41,278
Ethnicity: White/Anglo	93%	79.2%	91.3%

[1]The population demographics are shown by the two counties included in the study watershed due to the readily available Census data.

[2]Mean income for survey respondents was 4.8, which is between the US$35,000-49,999 (4) and US$50,000-74,999 (5) ordinal response categories.

STUDY RESULTS

The extent of positive and negative judgments about water resource protection are presented with descriptive statistics and illustrative comments from interviews and surveys, which enrich and validate the quantitative findings. A few exemplary quotes from residents also elaborate on particular attitudinal judgments. The factor analysis results further reveal the dimensions of attitudes and the structure of individual judgments about resource protection.

The Nature and Extent of Attitudes

General attitudes toward protecting streams, lakes, and wetlands were extremely positive, with almost all respondents attaching great importance to water resource protection. The vast majority of residents attached substantial importance to the broad management goals evaluated, with negative judgments expressed by only 2-15% (Table 1). Water quality ranked extremely high in importance, with minimal negative judgments for protecting resources for both clean waterways and drinking water. Overall, biocentric goals (clean water, habitat projection) ranked higher than anthropocentric goals (recreation, flood management), likely due to focus on "protecting" water resources in this study.

Judgments about government were moderately negative (15-30% opposition). Local (city, county) agencies were most supported, followed by state efforts and then the regional government (Metro). Federal efforts were least supported among the four levels. Attitudes about government are confounded by the fact that some people responded generally, noting they are unfamiliar with specific efforts, while others responded by citing specific efforts, such as those of G. W. Bush. Similar to the regional government, one-fifth of respondents opposed efforts by nonprofit organizations. Among the entities evaluated, business efforts were the most opposed (by about half of respondents). Written comments reveal a strong perception among residents that businesses are primarily responsible for resource degradation, as in, "Big businesses are the main polluters of water."

Educational outreach and restoration were highly preferred as general policy options, more so than land acquisition, financial incentives, and regulations (6-8% opposition compared to 18-21%). Many respondents (nearly 7%) selected "don't know" for financial incentives, indicating a lack of understanding or perhaps a need for more information. Meanwhile, regulations garnered a fairly wide range of opposition (6-21%). The most negative attitudes were those about regulations generally, while specific restrictions on *how* development is designed (to minimize impacts) were most supported, followed by regulations on nonresidential (industrial and commercial) land. Respondents were somewhat more opposed to regulating residential land, in addition to restricting new construction, the removal of trees, and the plants allowed near water. According to an interview informant, residents' concerns about regulations often subside when they are informed about the details of their application (for example, whether they will be allowed to maintain their gardens).

Economic measures received the widest range of negative attitudes, with 5-53% opposition. The vast majority of respondents expressed support for current or higher levels of funding through fines on land use violations and taxes on polluting products, while support for fees on new development was slightly lower. These attitudes represent a "make the polluter pay" attitude, as emphasized in written comments such as, "I believe the polluter should pay, and I do mean everyone—farmers, businesses as well as the city..."

Moderate opposition to voter-approved bonds is likely due to recent land acquisition measures, since polls have indicated a weakening of support for new land purchases and heightened support for improvements on land previously acquired. About one-third of respondents opposed paying for resource protection personally, with several noting they already pay through water bills and taxes. Indeed, these funding mechanisms were widely opposed (by about half of respondents), partly due to the "unfair" burden on residents, which was expressed as follows: "The taxpayers are always asked to pick up and pay when corporations don't [or] won't."

In sum, residents expressed a wide range of negative attitudes (2-53%) toward protecting water resources in the Portland area, with the greatest opposition toward businesses, taxes, and measures aimed at residents. Meanwhile, residents highly supported water quality goals, voluntary approaches, and regulatory and economic measures targeting businesses, who are commonly seen as "the main polluters" of water. The sentiments and patterns highlighted in this description of attitudes are further underscored by the factor analysis results presented in the next section.

The Dimensions of Attitudes

According to standard statistical criteria (Kim and Mueller, 1978), the *factor matrix* identified one dominant factor that accounted for 40% of the variation in the data (Table 3). The highest loadings were judgments about regulatory and government efforts. A few attitudinal variables had loadings below the minimum (0.35) criterion, and others had higher loadings on subsequent factors compared to the primary one. Specifically, the anthropocentric goals—that is, protecting resources for drinking water, recreational enjoyment, and flood management—did not load significantly onto the overall attitudinal factor. Meanwhile, biocentric goals (wildlife habitat and *in situ* water quality) had relatively high loadings on the primary dimension and also comprised a separate dimension. In addition, judgments about income and property taxes loaded more highly onto a distinct, subsequent factor. Beyond the primary dimension, six factors (with Eigenvalues greater than one) accounted for an additional 25% of the variation in residents' judgments, with a total of two-thirds of the variation explained by seven factors.

In the *pattern matrix*, the first dimension represents attitudes toward specific types of land use restrictions. Judgments about regulations generally had a relatively low but significant loading (Table 3). Additional judgments about regulations loaded onto another factor, distinguishing attitudes about specific types of near-water restrictions from those about the regulation of private land.

The funding mechanisms most directly impacting residents, particularly income and property taxes, comprised the second factor in the pattern matrix. Attitudes about funding resource protection through water and sewer bills also loaded onto this factor. As stated earlier, these were among the most opposed measures for resource protection.

Judgments about the entities involved in resource management loaded onto the third factor, with high loadings for attitudes toward government at the local, state, and federal levels. Attitudes about business and nonprofit actors had relatively low loadings. A sub-dimension of the factor matrix further set apart attitudes toward for-profit entities, with positive loadings for business efforts and negative loadings for regulating industrial and commercial land.

TABLE 3. Factor Analysis Results: Significant Loadings for Dimensions of Attitudes About "Protecting" Surface Water Resources.

Individual Attitude Variables	Primary Matrix: Overall Attitudes (Factor 1)	Pattern Matrix: Attitude Dimensions by Factor Number and Descriptive Labels	
General importance	0.663	0.308^	**(4)** Protection Goals (Biocentric)
Drinking water quality	-	0.449	
Human use/enjoyment	-	0.489	
Flood management	-	0.533	
Wildlife habitat	0.679	0.558	
Clean streams/lakes	0.602	0.732	
Financial incentives	0.472	0.691	**(5)** Voluntary Efforts
Purchasing land	0.574	0.634	
Education and outreach	0.582	0.426	
Regulating parks/open spaces	0.737	0.361	
Expressed willingness to pay	0.629	0.343^	
Restoration projects	0.747	0.270^	
Federal government	0.613	0.725	**(3)** Management Entities (Government)
State government	0.702	0.717	
Local government	0.713	0.617	
Regional government	0.737	0.499	
Nonprofit organizations	0.666	0.414	
For-profit businesses	0.390	0.583	
Regulations (generally)	0.764	0.422	(1) Land Use Restrictions
Restricting new development	0.686	0.720	
Restricting how designed	0.783	0.715	
Restricting tree removal	0.728	0.835	
Restricting plants allowed	0.769	0.883	
Regulating residential areas	0.750	0.401	**(6)** Private Regulations
Regulating agricultural areas	0.755	0.473	
Regulating commercial areas	0.768	0.462/−0.375	**6/7**
Regulating industrial areas	0.707	0.455/−0.408	
Fees on new development	0.579	−0.414	**(7)** Polluters (Businesses) Pay
Taxes on polluting products	0.525	−0.640	
Fines for land use violation	0.525	−0.748	
Voter-approved bonds	0.432	0.340^	**(2)** Residents Pay
Charges on water/sewer bills	0.440	0.392	
Income taxes	0.458	0.764	
Property taxes	0.481	0.787	

Notes: The factor analysis employed principal axis extraction with Direct Oblimin rotation. Loadings >0.35 are presented, with emphasis on high (>0.6) values in the factor labels and discussion of findings. For variables that did not load significantly onto any factors, the highest value is presented and marked with a carrot(^).

The fourth dimension represents attitudes about broad resource protection goals, with the highest loadings for clean streams, lakes, and wetlands, followed by wildlife habitat and flood management. In the primary factor matrix, a separate dimension represented biocentric objectives (wildlife habitat and clean waterways). Judgments about flood management may reflect biocentric as well as anthropocentric values given recent focus on wetland restoration for flood mitigation purposes. Anthropocentric attitudes about drinking water and recreation goals had relatively low loadings on this factor, which reflects judgments about the general importance of resource protection, especially for biocentric purposes.

Voluntary market-based measures including land acquisition and financial incentives dominated the fifth factor, which also had significant but low loadings for educational outreach and regulation of public land. The latter implies self-regulation of the government, largely maintaining the voluntary nature of resource protection as far as residents are concerned.

Attitudes toward regulating different land uses loaded separately onto the sixth factor. In particular, judgments about regulating residential, agricultural, industrial, and commercial land reflect judgments about restricting private land uses, with insignificant loadings for regulating public parks and open spaces.

For the seventh factor of the pattern matrix, judgments about regulating commercial and industrial land had negative loadings, with high positive loadings for attitudes toward funding resource protection through taxes on polluting products, fines on land use violations, and fees on new development. These economic measures represent efforts to "make polluting businesses pay," a common sentiment in residents' written comments.

In short, the attitudinal dimensions in the pattern matrix highlighted distinct judgments about: (1) specific land use restrictions near waterways; (2) residents' financial contributions; (3) management actors, especially government; (4) resource protection goals, especially biocentric objectives; (5) voluntary policies and programs; (6) regulation of private land; and (7) strategies that make polluters (particularly businesses) pay (Table 3).

The Structure of Attitudinal Dimensions

The judgments examined in this study are grouped by the seven dimensions of attitudes derived from the factor analysis, which are further organized by three overarching attitude objects – the goals, entities, and strategies involved in resource management. The empirical structure of attitudes is illustrated graphically as a hierarchy of judgments (Figure 1), with Cronbach's alpha values noted to indicate their reliability (Spector, 1992). The ideal criterion is an alpha (α) greater than 0.7, although 0.5 or higher is an acceptable minimum (Nunnally, 1967; Albrecht *et al.*, 1982). Evaluating public attitudes with multiple internally consistent judgments is imperative because environmental views are too complex to be adequately addressed by single measures (Van Liere and Dunlap, 1980; Whittaker *et al.*, 2005). Thus, the attitudinal scales are compared statistically herein to identify significant differences in the strength of judgments about particular aspects of resource protection (Table 4).

First, attitudes toward management goals ($\alpha = 0.77$) are classified by biocentric and human-centric purposes ($\alpha = 0.82$ and 0.62, respectively). In this study, biocentric judgments were significantly more positive than anthropocentric ones (Table 4). Although attitudes about "*protecting*" water resources appear largely driven by biocentric interests, future research should further distinguish between biocentric and anthropocentric goals in a variety of resource *management* contexts. For activities fulfilling both objectives (such as flood management efforts), clarifying the tactics (such as restoring wetlands, developing parks, or building levees) and the associated outcomes (for example, habitat improvements, recreational opportunities, or floodwater mitigation) is important for understanding attitudes in relation to the diverse means available for achieving single or multiple goals.

Second, judgments about entities ($\alpha = 0.85$) are distinguished by governmental/nongovernmental actors, with higher support and enhanced reliability for attitudes toward government ($\alpha = 0.87$) compared to the other entities examined ($\alpha = 0.66$). Future work should evaluate attitudes toward additional nongovernmental groups as well as the efforts of individuals, such as residents themselves. Focusing on attitudes toward specific government agencies or projects is also worthwhile, since attitudes appear contingent upon such details and general attitudes may mask variation in judgments. Moreover, because attitudes for particular levels of government differ, the patterns among individual judgments (Table 1) should be examined along with the composite attitudinal scales.

Third, attitudes toward water resource management strategies ($\alpha = 0.92$) are distinguished by voluntary ($\alpha = 0.87$), regulatory ($\alpha = 0.92$), and economic ($\alpha = 0.85$) approaches. Attitudes about regulatory and economic strategies encompass two subdimensions. In particular, support for specific types of restrictions ($\alpha = 0.89$) was significantly higher than for regulating private land ($\alpha = 0.88$), whereas support for economic measures targeting polluting businesses was substantially higher than those targeting residents ($\alpha = 0.75$ for both). Interestingly, support for regulatory approaches was significantly stronger than for voluntary efforts (Table 4), perhaps due to concerns about the funding sources for voluntary programs such as those involving financial incentives.

The only judgments spanning the three subcategories of strategies (economic, regulatory, and voluntary) were those pertaining to economic regulations aimed at polluters (see Figure 1). Specifically, support for funding resource protection with fees on detrimental activities combined with regulations on for-profit land uses to reflect a "polluting businesses should pay" mentality ($\alpha = 0.80$).

In sum, judgments about water resource management reflect dimensions along which attitudes vary in meaningful ways. Evaluating composite scales for specific attitudinal dimensions assists with reliable assessments of public opposition and policy preferences. As reported elsewhere (Larson, 2005; Larson and Santelmann, 2007; Larson and Lach, 2008), additional analyses of four attitudinal dimensions pertaining to resource protection—management *goals*, *government* entities, *regulatory* approaches, and *eco-*

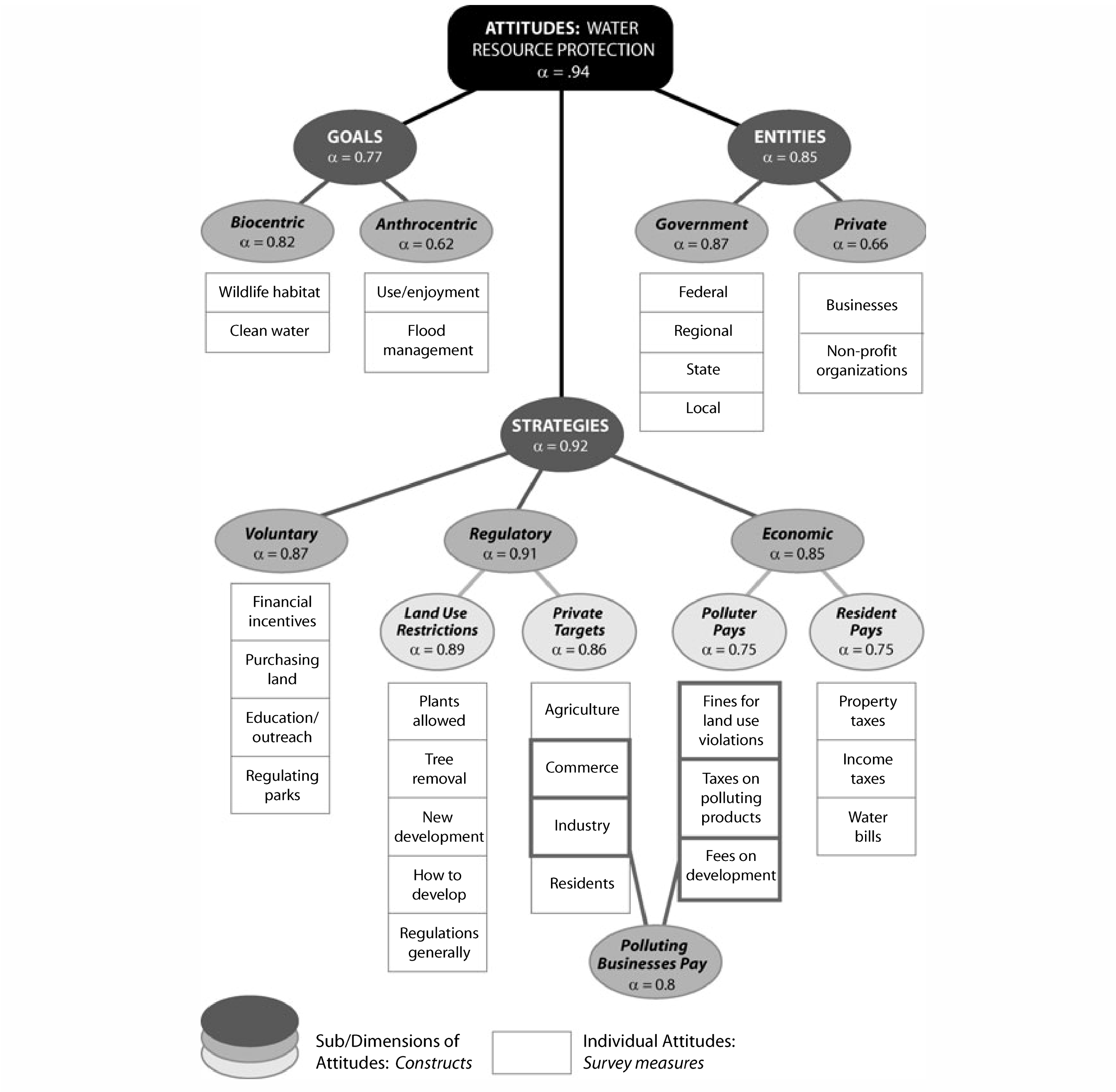

FIGURE 1. The Reliability and Hierarchical Structure of Attitudinal Dimensions.

nomic measures—underscored their validity. A regression analyses showed, for instance, that while ecological worldviews were most critical in explaining attitudes toward broad resource protection goals, political orientation was especially crucial for attitudes about government. The specific ecological and political beliefs influencing attitudes varied across dimensions as well; ideological belief in private property rights was central for judgments about regulations, whereas beliefs about government intervention in the free market were critical for economic opposition. In addition, proximity to surface water was differently linked to the attitudinal dimensions, with a significant negative association for regulatory support and, controlling for adjacency, a positive association with economic support (Larson and Santelmann, 2007). As a whole, this empirical study firmly establishes the reliability and validity of significant dimensions of environmental attitudes while also providing insights for future research.

TABLE 4. Statistics for Attitude Scales: Reliability and Tests of Differences.

Attitude Scales[1]	Cronbach's Alpha[2]	No. of Items	Scale Mean	SD.	Max. (min = 1)	N
Goals (t = 17.07**)						
Biocentric	0.82	2	1.72	0.910	6	809
Anthropocentric	0.62	2	2.30	1.025	6	809
Entities (t = 7.87**)						
Government	0.87	4	2.44	1.206	6	743
Nongovernment	0.66	2	2.82	1.427	6	723
Strategies[1] (t = 3.80**)						
Voluntary	0.72	4	2.10	0.960	6	780
Regulatory (t = 3.45*):	0.91	9	1.99	0.998	6	784
Private Land	0.88[a]	4	1.90	1.021	6	759
Restricting Uses	0.89	5	2.04	1.126	6	782
Economic (t = 41.81**):	0.78	6	1.91	0.462	3	780
Polluters Pay	0.75	3	1.47	0.542	3	772
Residents Pay	0.75[b]	3	2.35	0.520	3	765

Notes: Attitudes scales are composite variables equal to the average of the individual survey responses (see Figure 1 for items included in each scale).

[1]For judgments about goals, entities, and strategies, paired statistical tests of differences (significant at $p < 0.001$** and $p < 0.01$*) were conducted for the subscales. Due to the unique scale used to evaluate judgments about funding mechanism, paired tests of differences were only conducted for the voluntary–regulatory strategies as well as the two subdimensions of economic attitudes. The grey-shading serves as a reminder that the economic measures included a shorter scale (see Appendix).

[2]A few alpha values were higher with items deleted, specifically: [a] indicates without residential land, regulating private (for-profit) land = 0.92, and [b] indicates without water bill charges, residents' willingness to pay (taxes) = 0.81.

ASSESSING DIMENSIONS OF ENVIRONMENTAL ATTITUDES

The approach outlined here for evaluating environmental attitudes articulates important dimensions along which judgments vary (for a detailed theoretical review of the framework, see Larson, forthcoming). A critical advantage of this approach is the ability to evaluate multifaceted environmental attitudes across unique problems, populations, places, and time periods. Standard approaches have been constructively employed in the past to illustrate changing commitment to the NEP over time (Dunlap and Van Liere, 1978; Dunlap *et al.*, 2000) and differing values across cultural groups and nations (Schwartz, 1994; Schultz and Zelezny, 1999). My approach maintains the benefits of such comparative research. Instead of offering a fixed measurement scale, however, the conceptual framework allows for the evaluation of a nearly infinite number of possible judgments about diverse attitude objects (goals, entities, and strategies) depending upon applied resource management issues and basic research questions or objectives.

An Emergent Framework for Future Assessments

The conceptual framework is organized around assessing attitudes toward the primary *goals*, the *entities* involved, and the *strategies* undertaken for resource management. Attitudes toward resource planning and protection efforts can bye evaluated not only by assessing judgments toward each element separately, but also by combining various goals, entities, and strategies to reflect real-world approaches underway or under consideration, in addition to conceptually meaningful constructs such as the "make the polluter pay" mentality.

Beyond examining attitudes about particular goals, entities, and strategies, the framework may be applied to different *resource domains* or *targets*. While this study focused on "protecting" water resources such as streams and wetlands, other studies might explicitly address flood management, habitat restoration, pollution mitigation, or other resource management domains. Additional research might also distinguish between resource targets, such as different types of water (surface streams *vs*. ground water aquifers, natural *vs*. human-made lakes) or particular water bodies (for example, Johnson Creek *vs*. the Willamette or Columbia Rivers), since such distinctions may very well influence attitudinal judgments.

With respect to management goals, attitudes should be evaluated in relation to associated values including biocentric–anthropocentric orientations and personal (individual)–social (collective) interests. By combining these two dimensions, attitudinal responses are likely to differ toward the following types of management objectives: human-centered goals that satisfy personal self-interests, human-centered goals that serve societal benefits beyond selfish interests, biocentric goals that entail personal interests, and biocentric goals that entail altruistic values. In this study, the focus was primarily on the biocentric–anthropocentric dimension, with attitudes about protecting streams and wetlands eliciting primarily biocentric attitudes. Yet selfish interests also appeared to underlie the tendency for residents to support efforts aimed at nonresidential activities, with relatively high opposition to those activities directly impacting them (such as taxes and regulations on residential land). Future research should carefully evaluate attitudes relative to personal impacts (for example, whether residents live in a flood prone area or have particular business interests), in order to determine the influence of utilitarian self-interests (due to the positive and negative impacts of specific activities) compared to broad-based values and orientations.

Attitudes also vary based on the entities initiating resource management, including nongovernmental groups and government at nested geographic scales. An important distinction concerning the management entities is between the *actors* initiating or implementing activities *vs*. the *targets* impacted or involved (for example, in regulations or fines). Combing the nongovernment and individual-group dimensions, attitudes may be assessed toward: the government as a whole (group-level) including different levels or particular agencies, nongovernment organizations including different types of businesses and nonprofit groups, individual government representatives such as particular agency personnel or elected officials, and nongovernment individuals such as residents or representatives of organized interest groups. The individual-group distinction is important because attitudes depend on the particular entities evaluated (such as G. W. Bush specifically *vs*. the federal government generally), in addition to social-political beliefs pertaining to top-down organized management *vs*. bottom-up collective actions by individuals. Attitudes involving individual research participants themselves, relative to other people or entities, are also critical to consider given underlying selfish–altruistic values and, as found in this study, the propensity for residents to attribute resource degradation to other, nonresidential actors. With the rising number of nongovernmental civic organizations (Allan, 2005), research focusing on attitudes toward nonprofit groups with diverse missions and tactics is also worthwhile.

The final element of the framework focuses on the *strategies* employed in resource protection and planning – that is, the policy tools, program types, and management activities. Because attitudes toward different approaches differ based on social values and political beliefs, the framework characterizes voluntary–regulatory and noneconomic-based strategies, as follows: regulations lacking an economic or market basis, economic-based restrictions such as fines for violating regulatory standards, voluntary economic measures including incentives and other market-based approaches, and voluntary noneconomic approaches including restoration efforts, educational activities, marketing campaigns, and nonmonetary incentives (such as those acknowledging stewardship). Further investigation of attitudes toward a variety of incentive-based and economic-based approaches is especially warranted due to the shift toward decentralized management and attention to market-oriented policies in recent years (Lant, 1998; Allan, 2005).

Applying the Framework

The framework outlines dimensions of attitudes to be evaluated and compared broadly in future studies, such that the approach may be flexibly employed to meet specific project objectives. In order to facilitate comparative assessments using this approach, similar presentations of findings are recommended, specifically, the percent of respondents expressing negative (or positive) attitudes for individual judgments (Table 1), factor loadings for the individual judgments grouped by attitudinal dimensions (Table 3), and descriptive statistics and Cronbach's alpha values indicating the nature and reliability of composite attitude scales (Table 4). The presentation of judgments as percent-negative permits comparisons regardless of the response scales employed in specific studies. To allow direct comparisons, the publication of verbatim survey questions (see Appendix) is incredibly useful.

Quantitative and qualitative research methods are highly recommended to validate findings about complex, multifaceted attitudes. Preliminary interviews with local informants assist with identifying current and potential management approaches toward which attitudes are evaluated, thereby maximizing the policy relevance of such studies. Meanwhile, quantitative survey measures gauge the acceptance of specific initiatives among targeted populations, facilitating numerical comparisons and the generalization of empirical findings across studies. Finally, broad conceptual comparisons among different studies will advance knowledge and theoretical explanations for multidimensional environmental attitudes.

CONCLUSION AND RECOMMENDATIONS

This study illustrates how residents' judgments about water resource protection in metropolitan Portland, Oregon vary by the objectives, actors, and policy strategies employed, such that attitudes reflect biocentric value orientations, beliefs about government and businesses, and ideological preferences for voluntary, regulatory, and economic approaches. Judgments toward regulations and government were critically important for understanding environmental attitudes, reflecting the importance of underlying social-political values and beliefs. Meanwhile, self-interests were evident in judgments about funding mechanisms targeted at residents' own pocket books. In addition to distinct judgments representing a personal willingness (or lack thereof) to bear the costs of resource protection, residents expressed substantially higher economic support for measures emphasizing the "make the polluter pay" principle. Such attitudes were expressly linked to businesses, who were largely seen as the entities responsible for resource degradation. Residents also expressed high support for regulation of industry and commerce, and judgments were distinguished by specific types of restrictions and regulation of private land.

Generally, residents appear to become supportive of regulations as they understand the specifics of their application. Resource planners and managers might therefore ease concerns and garner support for regulations by detailing their implications. Management efforts that reward stewards and penalize degraders of resources are also likely to earn support compared to others, whether through economic, regulatory, or other approaches. Yet support for voluntary policy tools such as land acquisition may be a function of their funding mechanisms, with the highest support for initiatives that "make the polluter pay." Given heightened economic support for resource protection among those who live near streams or use resource areas for recreational purposes (Larson and Santelmann, 2007), targeting mandatory or voluntary funding measures at the people who value and benefit most from resource protection is another viable management strategy.

Regardless of policy or program objectives, emphasizing the multiple values of resource management is likely to increase support for initiatives among diverse people who may be biocentric or anthropocentric in their value orientations. For programs aimed at protecting fish and wildlife, this might involve allowing access to acquired open spaces for recreational purposes or designing habitat areas with "cues of care" to enhance both the ecological and cultural sustainability of public lands (Nassauer *et al.*, 2001). Locally based programs that connect residents to resources are indeed of utmost importance, and partnerships with local governments and entities may reduce opposition to initiatives coordinated at broader scales of governance.

A critical challenge for programs that target residents may be the perception that particular actors, such as businesses, are primarily responsible for resource degradation. Consequently, balanced programs involving a variety of land and resource users might be necessary to overcome the view that programs targeting households are unfair. Effectively illustrating the significant impacts that household activities have on resources, especially in urban and urbanizing watersheds, is essential to address in endeavors involving or targeting residents.

Based on the empirically and theoretically derived dimensions of attitudes, the conceptual framework presented herein is intended to guide future evaluations of judgments about a variety of goals, entities, and strategies involved in water resource management. As focus is placed on various elements of the framework, consideration should be given to the values and beliefs influencing attitudinal dimensions and how these vary across people, places, and problem domains. While facilitating comparative research to advance knowledge of attitudes, the framework is sufficiently flexible to explore unique research questions and allow attention to specific geographic contexts. By understanding multidimensional environmental attitudes as important elements of sustainable resource use and protection, research can inform and enhance the social acceptability and political feasibility of water and environmental management regimes.

APPENDIX: SURVEY MEASURES OF ATTITUDES

All of the attitudinal items on the survey are included verbatim below. Response labels are in capital letters. "No opinion" and "don't know" options were offered for all of the below questions. Most of the survey questions had anchored response options at the top with no middle labels, with each item having evenly dispersed numbers and ellipses in-between for respondents to circle their preferred option. The exceptions were question numbers four and six below, which had the noted labels for each of the numbered response options.

1. Overall, how important do you think it is to protect the condition of water resources in the greater Portland metropolitan area? (1) VERY IMPORTANT—(6) NOT AT ALL IMPORTANT

2. Please mark on the scales provided how important you think it is to protect the condition of water resources in the greater Portland metropolitan area

for each of the following purposes? (1) VERY IMPORTANT—(6) NOT AT ALL IMPORTANT

a. Drinking water quality

b. Clean streams, lakes, and wetlands

c. Flood management

d. Fish and wildlife habitat

e. Public use and enjoyment

f. Other (specify):_____________

3. To what degree do you support or oppose efforts by each of the following groups to protect the condition of water resources in the greater Portland metropolitan area? (1) STRONGLY SUPPORT—(6) STRONGLY OPPOSE

a. Local government (city, county)

b. Regional government (Metro)

c. State government

d. Federal government

e. Nonprofit organizations

f. Businesses (for profit)

g. Other (specify):____________

4. Are you personally willing to pay for efforts aimed at protecting the condition of water resources in the greater Portland metropolitan area? (1) NO—(2) PROBABLY NOT—(3) PROBABLY—(4) YES

5. To what extent do you support or oppose the government using each of the following options to protect the condition of water resources such as streams, rivers, lakes, and wetlands in the greater Portland metropolitan area? (1) STRONGLY SUPPORT—(6) STRONGLY OPPOSE

a. Financial incentives

b. Public outreach and education

c. Purchasing land (from willing sellers)

d. Regulations on how land is used/developed

e. Restoration projects to improve the condition of water resources

f. Other (specify):_____________

6. Please indicate whether or not you support using the following funding types to pay for programs aimed at protecting the condition of water resources in the greater Portland metropolitan area? (1) SUPPORT HIGHER LEVELS—(2) SUPPORT CURRENT LEVELS—(3) DON'T SUPPORT (OPPOSE)

a. Charges on water/sewer bills

b. Fees on new development

c. Fines collected from violations of land use regulations

d. Income taxes

e. Property taxes

f. Taxes on polluting products such as pesticides

g. Voter approved bond measure

h. Other (specify):____________

7. Mark the degree to which you support or oppose government regulations to protect the condition of water resources for each of the following types of land in the greater Portland metropolitan area. (1) STRONGLY SUPPORT—(6) STRONGLY OPPOSE

a. Residential (housing) areas

b. Commercial areas (stores, offices)

c. Industrial areas

d. Agricultural areas

e. Public parks and open spaces

f. Other type of land: __________

8. To what degree do you support or oppose the following types of government regulations for protecting the condition of water resources in the greater Portland metropolitan area? (1) STRONGLY SUPPORT—(6) STRONGLY OPPOSE

a. Restricting new construction near water resources.

b. Restricting how new development is designed and constructed (to minimize impacts to water resources).

c. Restricting the removal of trees or other vegetation near water resources.

d. Restricting the types of vegetation that can be planted near water resources.

9. Explain below why you support or oppose government efforts to protect the condition of water resources such as streams, rivers, lakes, and wetlands in the greater Portland metropolitan area. Please note any views you have on specific water resource programs in the Portland metropolitan region.

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STREAMBANK SOIL AND PHOSPHORUS LOSSES UNDER DIFFERENT RIPARIAN LAND-USES IN IOWA[1]

George N. Zaimes, Richard C. Schultz, and Thomas M. Isenhart[2]

ABSTRACT: Phosphorus and sediment are major nonpoint source pollutants that degrade water quality. Streambank erosion can contribute a significant percentage of the phosphorus and sediment load in streams. Riparian land-uses can heavily influence streambank erosion. The objective of this study was to compare streambank erosion along reaches of row-cropped fields, continuous, rotational and intensive rotational grazed pastures, pastures where cattle were fenced out of the stream, grass filters and riparian forest buffers, in three physiographic regions of Iowa. Streambank erosion was measured by surveying the extent of severely eroding banks within each riparian land-use reach and randomly establishing pin plots on subsets of those eroding banks. Based on these measurements, streambank erosion rate, erosion activity, maximum pin plot erosion rate, percentage of streambank length with severely eroding banks, and soil and phosphorus losses per unit length of stream reach were compared among the riparian land-uses. Riparian forest buffers had the lowest streambank erosion rate (15-46 mm/year) and contributed the least soil (5-18 tonne/km/year) and phosphorus (2-6 kg/km/year) to stream channels. Riparian forest buffers were followed by grass filters (erosion rates 41-106 mm/year, soil losses 22-47 tonne/km/year, phosphorus losses 9-14 kg/km/year) and pastures where cattle were fenced out of the stream (erosion rates 22-58 mm/year, soil losses 6-61 tonne/km/year, phosphorus losses 3-34 kg/km/year). The streambank erosion rates for the continuous, rotational, and intensive rotational pastures were 101-171, 104-122, and 94-170 mm/year, respectively. The soil losses for the continuous, rotational, and intensive rotational pastures were 197-264, 94-266, and 124-153 tonne/km/year, respectively, while the phosphorus losses were 71-123, 37-122, and 66 kg/km/year, respectively. The only significant differences for these pasture practices were found among the percentage of severely eroding bank lengths with intensive rotational grazed pastures having the least compared to the continuous and rotational grazed pastures. Row-cropped fields had the highest streambank erosion rates (239 mm/year) and soil losses (304 tonne/km/year) and very high phosphorus losses (108 kg/km/year).

(KEY TERMS: riparian areas; streambank erosion; soil and phosphorus losses; best management practices; grazing practices; nonpoint source pollution.)

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INTRODUCTION

Phosphorus has been identified as the limiting nutrient for eutrophication of many surface waters (Daniel *et al.*, 1998), while sediment is the number one water quality problem in the United States (U.S.) (Simon and Darby, 1999). Because phosphorus is typically transported with sediment (David and Gentry, 2000), reducing sediment delivery to streams can reduce both of these pollutants.

Streambank erosion has been suggested as a major contributor of sediment to streams. Sekely *et al.* (2002) estimated that streambank erosion in a Minnesota stream contributed 30-45% of the sediment load to streams, while Odgaard (1984) and Schilling and Wolter (2000) estimated a higher contribution of 45-50% in several Iowa streams. In other regions of the U.S. (Simon *et al.*, 1996) and other countries (Kronvang *et al.*, 1997), the contribution was estimated to be up to 80-90%. Very few studies have estimated streambank erosion contributions to stream total phosphorus load (Sekely *et al.*, 2002). In Minnesota, Sekely *et al.* (2002) estimated that only 7-10% of the total phosphorus in the stream was from streambank erosion, while in Illinois, Roseboom (1987) estimated it to be more than 55%. In Denmark, Kronvang *et al.* (1997) estimated streambank erosion to contribute more than 90% of the stream total phosphorus load.

Decreased streambank stability in many cases is the result of reduced vegetation cover that decreases root length and mass in the soil (Dunaway *et al.*, 1994). Livestock overgrazing is one land-use practice that can dramatically impact vegetation cover. Belsky *et al.* (1999) reported many studies that have shown livestock grazing reducing streambank stability in the western U.S. Rotational and intensive rotational grazing are slowly replacing traditional continuous grazing in Iowa because they maintain more vegetative cover providing better utilization of pasture forages, increased profitability and are generally considered more environmentally friendly (USDA-NRCS, 1997a). In the rotational and intensive rotational grazing, the pastures are divided in small sections (paddocks) and livestock are moved from one paddock to the next providing short intensive grazing pressure in a paddock followed by long periods of rest and recovery. The result is more complete utilization of the available forage, with time for that forage to regrow and maintain healthy and strong root systems. While many studies on the influence of intensive rotational and rotational grazing on stream ecosystems have been conducted in the western U.S, very few have been conducted in the Midwest (Lyons *et al.*, 2000). The objective of this study was to compare streambank erosion along reaches with different riparian land-uses, with a specific focus on grazing practices. For this comparison, six different streambank erosion variables were used: (1) erosion rate, (2) erosion activity, (3) maximum pin plot erosion rate, (4) percentage of severely eroding bank length, (5) soil, and (6) phosphorus loss per unit of stream length. Based on the potential intensity of the land-use on the riparian vegetation and streambanks, we hypothesized that the streambank erosion would be as follows: row-cropped fields > continuous pastures > rotational pastures > intensive rotational pastures > pastures where cattle were fenced out of the stream > grass filters > riparian forest buffers. This study complemented earlier research conducted by the authors (Zaimes *et al.*, 2004, 2006) that examined only three different land-uses, riparian forest buffers, continuous pastures, and row-cropped fields, along one single stream in central Iowa.

STUDY AREA

Iowa's natural vegetation has been altered more than any other state in the U.S. In the last 150 years, 99.9% of the tall-grass prairies were plowed, 95% of the wetlands were drained, and 70% of the forests were cut (Whitney, 1994). In their place, more than 90% of the land is now in annual row-crops and grazed pastures (Burkart *et al.*, 1994).

Stream reaches selected for this study were located in northeast and southeast Iowa because these are two major livestock grazing regions in Iowa. The Iowan Surface and the Paleozoic Plateau are the major landforms in northeast Iowa (Prior, 1991). The Paleozoic Plateau has narrow valleys in sedimentary rock with almost no glacial deposits, and because of the shallow limestone near the surface, there are numerous caves, springs, and sinkholes. The Iowan Surface is dominated by gently rolling terrain created by material loosened and moved by many weathering events caused by conditions during the last glaciation. The Southern Iowa Drift Plains landform, in southeast Iowa has many gullies, creeks, and rivers, with steeply rolling hills and valleys (Prior, 1991). Streambank erosion has deepened channels into glacial material deposited 500,000 years ago while a mantle of loess covers the slopes and hills. In addition, stream reaches in central Iowa were also used with one of these in the *Bear Creek National Restoration Demonstration Watershed* where a previous study had been conducted (Zaimes *et al.*, 2004, 2006). Central Iowa lies on the Des Moines Lobe landform that has subtly rolling terrain with some broad curved bands or ridges, knobby hills, and irregular ponds and wetlands resulting from the most recent glaciation in Iowa (Prior, 1991).

Riparian Land-Uses

The riparian land-uses of interest were annual row-cropped fields (RC), continuous (CP), rotational (RP), intensive rotational (IP) grazed pastures, pastures where the cattle were fenced out of the stream (FP), grass filters (GF), and riparian forest buffers (RF). Besides riparian land-use, the major criteria for selecting study reaches were as follows: (1) having lengths >300 m with the same land-use on both streambanks, (2) located along first- to third-order streams (Strahler, 1957), (3) channels in the widening stage (Stage III) of the channel evolution model (Schumm *et al.*, 1984), and (4) owned by private farmers. The focus was on low-order streams because they are in closest contact with their adjacent hillslopes, and therefore can contribute a significant portion of the sediment to larger streams. Low-order streams contribute 30-50% of the sediment to the Illinois River (Johnson, 2003). Working on private farms allowed evaluation of actual land-use management as practiced by farmers in the different regions. It was also felt that working with private farmers would make it easier to convince other farmers to change their practices by demonstrating results on neighboring farms.

Over a six-month period, more than 70 landowners and 120 reaches were visited, to eventually select 30 study reaches. It was not possible to find suitable reaches for all the riparian land-uses in every region. The riparian land-uses and the number of reaches of each riparian land-use in each region are presented in Table 1. The slopes of the stream channels for all reaches were less than 2%. In the northeast and southeast region, the watershed area above each reach was <52 km^2 while in the central region the area was <78 km^2. The hillslopes above all of the riparian areas were dominated by agricultural row-crop fields with some pastures and homesteads and occasional small pockets of forests.

Suitable reaches with RC adjacent to the streambanks were found only in the central region. Corn (*Zea mays* L.) and soybean(s) [*Glycine max* (L.) Merr.] were the annual row-crops, grown in alternating years. These reaches typically had a narrow strip (<4 m) of grasses and/or annual weeds along the streambanks, although many of the row-crops were grown right up to the streambank edge.

All pastures of this study were grazed by beef cattle and were dominated by vegetation consisting of cool-season grasses and forbs such as Kentucky bluegrass (*Poa pratensis* L.), tall fescue (*Festuca arundinacea* Schreb.), reed canary grass (*Phalaris arundinacea* L.), smooth bromegrass (*Bromus inermis* L.), orchardgrass (*Dactylis glomerata* L.), white clover (*Trifolium repens* L.), and red clover (*Trifolium pratense* L.). Each landowner started and ended grazing on different dates for all pasture practices, which led to different numbers of total grazing days. The CP were not divided into paddocks, and the cattle had full access to the stream during the entire grazing period. In the northeast and central region, grazing started in early May and ended in early November. In the southeast one of the CP reaches followed similar dates, while in the other two the cattle remained on the pastures throughout the year. Supplemental feeds (like hay) were supplied to cattle that grazed year-around. The grazing period for the IP and RP also ran from early May to early November in all regions. In the RP, the pastures were divided into two to three paddocks. Each paddock was grazed 15-30 days and rested for 30 days. In the IP, the pastures were divided into more than six paddocks and each paddock was grazed 1-7 days and rested for 30-45 days. Because the RP and IP practices have only recently been adopted by farmers with beef cattle in Iowa, study reaches were selected only if they had been converted from continuously grazed or row-crop agriculture for more than three years.

In the FP reaches, cattle had no access to the channel for at least three years. While this is a practice that might have great potential for decreasing streambank erosion, many cattle farmers in Iowa are reluctant to adopt it because the stream is the main water source for the cattle and because of the extensive and costly fence maintenance that may be required after flashy floods, which often occur in low-order streams of Iowa.

The selected GF reaches were vegetated by introduced cool-season grasses (USDA-NRCS, 1997c). The RF reaches were vegetated by zones of trees, shrubs, and warm season grasses (USDA-NRCS, 1997b). In one case, an existing natural forest along the streamside was used in lieu of a designed riparian forest buffer. Reaches for both of these land-uses were selected only if they had been established for at least five years. These two land-uses are the major conservation practices in riparian areas of Iowa and much of the Midwest.

METHODS

Rainfall Data

Rainfall data were used from the National Oceanic and Atmospheric Administration (NOAA) weather

TABLE 1. General Characteristics of the Riparian Land-Use Reaches in This Study (2001-2004).

Riparian Land-Use	Reach#	Series[1]	Average Bank Height (m)	Width/ Depth	Soil Texture[1]	Stocking Rate (AUM)	Precipitation (cm) Year 1[2]	Year 2[2]	Year 3[2]	Year 1-3
Central										
Row-cropped fields	2	Spillville-Coland complex	1.7	3.3	Clay loam, Loam	N/A	74	75	87	241
Continuous pastures	2	Coland, Colo, Spillville-Coland complex	1.7	2.7	Silt loam, Clay loam, Loam	16.3-23.5	74-91	63-75	79-87	241-246
Rotational pastures	2	Coland, Coland-Terrill complex	1.5	2.1	Clay loam	15.0-33.9	91	75	79	246
Grass filters	2	Spillville, Spillville-Coland complex	1.7	2.5	Clay loam, Loam	N/A	74	75	87	241
Riparian forest buffers	2[3]	Coland, Hanlon-Spillville and Spillville-Coland complexes	1.5	3.8	Clay loam, Loam	N/A	74	75	87	241
Northeast										
Continuous pastures	3	Dorchester, Radford, Otter-Ossian complex	1.5	3.6	Silt loam	15.6-22.5	62-92	52-64	106-107	233-250
Intensive rotational pastures	3	Dorchester, Dorchester-Chaeseburge-Viney and Dorchester-Chaeseburge complexes	1.2	3.6	Silt loam	9.6-19.5	92-134	52-59	95-108	242-300
Cattle fenced out of streams	2	Radford, Spillville	1.4	2.3	Silt loam	N/A	91-92	52-55	104-106	249-250
Riparian forest buffers	2	Colo-Otter-Ossian complex, Spillville	1.0	2.3	Silt loam, Loam	N/A	91-92	52-55	104-106	249-250
Southeast										
Continuous pastures	3	Nodaway, Nodaway-Cantril complex	1.8	1.4	Silt loam, Loam	15.4-22.6	56-86	54-64	74-86	185-235
Rotational pastures	2	Nodaway	1.9	2.0	Silt loam	12.9-29.3	76-86	59-64	81-86	216-235
Intensive rotational pastures	2	Nodaway, Nodaway-Cantril complex	2.1	1.5	Silt loam, Loam	7.6-12.7	86-93	54-78	74-100	215-271
Cattle fenced out of stream	1	Nodaway	1.9	1.4	Silt loam	N/A	86	78	100	271
Grass filters	2	Amana, Nodaway	1.8	2.1	Silt loam	N/A	76-86	55-59	74-81	215-216

Notes: AUM, animal unit month; N/A, not applicable.
[1]Soil Survey Geographic (SSURGO) (2004).
[2]Year 1: August 2001-2002; Year 2: August 2002-2003; Year 3: August 2003-2004.
[3]In this region, a natural forest along the stream was used as a riparian forest buffer reach.

station closest to each study reach (NOAA, 2002-2004a, 2002-2004b). Yearly rainfall data were correlated to yearly streambank erosion. For this study, precipitation was the best available variable to correlate to streambank erosion because discharge data were not available for any of the low-order streams. The flashiness of most first- to third-order streams in Iowa allows good seasonal correlation between precipitation and discharge, especially in spring and early summer.

Erosion Pins

Steel rods, called erosion pins, were inserted perpendicularly into the streambank (Wolman, 1959). Hooke (1979) recommended that one-third of the pin should remain buried so as not to get lost during a major erosion event. In addition, pins should not exceed 800 mm in length, to minimize interference with streambank erosion processes. A length of 762 mm was used in this study because erosion rates of up to 500 mm per erosion event had been witnessed in similar size streams (Zaimes *et al.*, 2004). A diameter of 6.4 mm was selected because it was small enough to cause minimum disturbance to the banks but large enough to not bend under most high discharge events (Lawler, 1993).

Erosion pins are well suited for measuring bank erosion rates for short-time scales and when high resolution is needed (Lawler, 1993). Resolution can be as high as 5 mm (Simon *et al.*, 1999). Accuracy, in this study, was increased even more because all pin measurements were collected by one operator (Couper *et al.*, 2002). Each erosion pin plot included two horizontal rows of five pins each. Pins within these rows were placed 1 m apart for a total length of 4 m. The streambank heights of the pin plots across all reaches varied from 1.6 to 2.2 m. To consistently place the pins in similar bank positions among the streambanks, the horizontal rows were placed at 1/3 and 2/3 of the height of the bank. Erosion pin plots with similar dimensions have been found to not influence streambank erosion processes (Lawler, 1993).

Severely eroding streambanks were the only ones selected for pin plot placement because these banks are the major source of sediment in streams (Beeson and Doyle, 1995). These banks are bare with slumps, vegetative overhang and/or exposed tree roots (USDA-NRCS, 1998). A preliminary field survey was conducted along each study reach, to identify all severely eroding banks and to record their locations on recent aerial photographs (scale 1:24,000). A random numbers table was then used to select five of those severely eroding banks in each reach to establish pin plots.

At installation, approximately 50 mm of the erosion pins were left exposed. Exposed pin lengths were measured once in the spring, summer, and fall of each year from August 2001 to August 2004. During the winter season, most pin plots could not be measured because they were frequently covered with snow and ice. The large numbers of measured pins (1,500 total), on 30 study reaches, with many frequent measurements over a three-year period resulted in a larger number of observations than found in most other studies that have used this method (Lawler, 1993).

Erosion pin measurements provided three of the six different variables used in this study to compare streambank erosion among riparian land-uses. These variables were streambank erosion rate, erosion activity, and maximum pin plot erosion rate. To estimate *erosion rate*, the previous measurement of the exposed erosion pin length was subtracted from the most recent one. When the difference was positive, the exposed pin measurement represented erosion; if it was negative then the pin measurement represented deposition. In contrast, *erosion activity* was the absolute value of the subtraction between the previous and the most recent pin measurement. Couper and Maddock (2001) suggested recording the *change* (absolute value) of erosion pin measurements, regardless of whether they represent erosion or deposition because they measure how active (unstable) the streambank is. The streambank erosion rate and activity for a given erosion pin plot were estimated by averaging the erosion rate and activity, respectively, of all the pins in the plot. The average streambank erosion rate and activity for a riparian land-use was estimated by averaging the erosion pin rate and activity, respectively, of all the plots in the specific riparian land-use for the entire measuring period. To estimate the *maximum pin plot erosion rate*, the erosion rate of the pin plot with the highest erosion rate within each riparian land-use in each region was used (Couper and Maddock, 2001). When a pin was completely lost during an erosion event, an erosion value of 600 mm was assumed (Zaimes *et al.*, 2006).

Severely Eroding Bank Survey

In August 2002, a second more detailed field survey of all the severely eroding banks was conducted. During this survey, the total length and average height for all severely eroding banks within each reach were measured. The height was estimated to the nearest 10 cm with a scaled height pole and measured at several points along the eroding streambank to calculate an average. With these measurements, the total length of severely eroding banks was estimated for each reach. By dividing the total length of the severely eroding banks for each reach by the total bank length for the reach (sum of the length on both sides of the channel), the *percentage of severely eroding bank length* (fourth variable) was estimated (USDA-NRCS, 1998). The percentage was used for comparison among riparian land-uses because each riparian land-use had a different stream reach length. In addition, the sum of the product of the average height and length for each severely eroding bank was used to estimate the severely eroding bank area within each reach. Total severely eroding area for each riparian land-use was determined as the sum of severely eroding bank areas within each reach of the land-use.

Streambank Soil and Phosphorus Losses

The product of streambank erosion rate, streambank soil bulk density, and severely eroding bank area for each riparian land-use was used to estimate its total soil loss from the streambanks. Streambank erosion rate was the average rate of all the pin plots in the riparian land-use. By multiplying the total soil loss by the average streambank soil phosphorus concentration in each riparian land-use, total phosphorus loss from streambanks was estimated. Streambank *soil and phosphorus loss per unit of stream length* (fifth and sixth variable, respectively) were estimated by dividing the streambank soil and phosphorus loss for each riparian land-use by its total stream reach length. This was necessary because each riparian land-use had a different total stream reach length. The bulk density and phosphorus concentrations estimates used were from a complimentary study the authors conducted (Zaimes *et al.*, 2008) that has not yet been published. In this complementary study, bulk density and phosphorus concentrations were estimated by collecting soil samples from severely eroding banks in each reach.

Data Analysis

The analysis of covariance in the Statistical Analysis System (SAS) was used to examine impacts of riparian land-use on streambank erosion rate and activity for each year and for all three years (SAS Institute, 1999). The sample size was the number of pin plots in each riparian land-use. Rainfall was used as a covariate in the above model because even in the same region, some riparian land-use reaches received different amounts of rainfall. Analysis of variance in SAS was used to compare percentage of severely eroding bank lengths and the model included regions and riparian land-uses. Difference were considered significant when the p-values <0.10. The p-value is the probability of how much evidence we have against the null hypothesis (Kuehl, 1999).

RESULTS AND DISCUSSION

Streambanks are never completely stable, and natural processes, including streambank eorison, channel migration, succession of riparian vegetation, are always occurring. However, in Iowa and the rest of the Midwest, agricultural land-use alterations to the riparian and hillslope areas of many watersheds have triggered changes in bank stability that have led to accelerated erosion that is considered unnatural. This study investigated the impact of different riparian land-uses on accelerated streambank erosion.

Erosion Pins

Over the entire three-year period, average erosion rates among land-uses ranked as follows: in the central region, RC > CP > GF > RP > RF; in the northeast region, CP > IP > FP > RF; in the southeast region, RP > CP > IP > FP > GF (Table 2). Average erosion activities among land-uses in all three regions ranked in the same orders as erosion rates except in the southeast region where the CP and RP activities were the same (Table 3). As expected, erosion activities for all riparian land-uses were higher than the respective erosion rates. The differences among the riparian land-uses based on the above rankings were not always significant. Specifically, in the southeast region there were no significant differences in erosion rate or activity among any of the riparian land-uses (Tables 2 and 3).

In the central region, the banks along RC had significantly higher annual and three-year average erosion rates and activities than those along the RF, GF, and RP banks (Tables 2 and 3). The three-year average erosion rates and activities for the CP banks were significantly higher than those of the RF banks. The CP banks also had significantly higher erosion rates/or activities than the RF banks during the last two years of the study, and the GF and RP banks in Year 3. The differences we saw in this region were expected, although even more significant differences among the land-uses were expected. In the northeast region, the CP and IP banks had significantly higher three-year average erosion rates and/or activities than the RF and FP banks (Tables 2 and 3). In Year 1, RF and FP banks had net deposition, while the

TABLE 2. Streambank Erosion Rates Under Different Riparian Land-Uses in Three Iowa Regions.[1]

Riparian Land-Use	Streambank Erosion Rates[2]									
	Year 1[3] (mm)	SD[4]	Year 2[3] (Mm)	SD[4]	Year 3[3] (Mm)	SD[4]	Sum Year 1-3 (mm)	SD[4]	Average Year 1-3 (mm/year)	SD[4]
Central										
Row-cropped fields	225 (74)	a	223 (59)	a	271 (37)	ab	717 (137)	a	239 (46)	a
Continuous pastures	79 (71)	a	128 (64)	ab	298 (40)	a	499 (133)	ab	166 (44)	ab
Rotational pastures	70 (73)	a	54 (75)	b	198 (44)	bc	313 (135)	bc	104 (45)	bc
Grass filters	87 (74)	a	66 (59)	b	168 (37)	c	319 (137)	bc	106 (46)	bc
Riparian forest buffers	54 (74)	a	4 (59)	b	83 (37)	c	139 (137)	c	46 (46)	c
Northeast										
Continuous pastures	151 (63)	a	184 (48)	b	137 (45)	b	512 (109)	a	171 (36)	a
Intensive rotational pastures	114 (65)	ab	98 (53)	ab	313 (39)	a	511 (130)	a	170 (43)	a
Cattle fenced out of streams	−25 (73)[5]	b	51 (65)	ab	24 (47)	c	67 (137)	b	22 (46)	b
Riparian forest buffers	−10 (73)[5]	ab	36 (65)	a	1 (47)	c	45 (137)	b	15 (46)	b
Southeast										
Continuous pastures	127 (61)	a	23 (50)	a	182 (40)	a	302 (125)	a	101 (42)	a
Rotational pastures	166 (72)	a	16 (59)	a	199 (39)	a	366 (136)	a	122 (45)	a
Intensive rotational pastures	59 (72)	a	55 (62)	a	169 (37)	a	281 (134)	a	94 (45)	a
Cattle fenced out of stream	42 (102)	a	−6 (102)	a	95 (55)	a	173 (209)	a	58 (70)	a
Grass filters	37 (72)	a	12 (61)	a	109 (46)	a	123 (143)	a	41 (48)	a

[1]The mean rainfall that each riparian land-use reach received was used as a covariate to estimate streambank erosion rate. In parentheses is the standard error.
[2]To estimate *erosion rate*, the previous measurement of the exposed erosion pin length was subtracted from the most recent measurement. When the difference was positive, the exposed pin measurement represented erosion; if it was negative the pin measurement represented deposition. Streambank erosion rate was the average rate of all the pin plots in the riparian land-use of a region.
[3]Year 1: August 2001-2002; Year 2: August 2002-2003; Year 3: August 2003-2004.
[4]SD, significant differences. In this column the different letters indicate significant differences (p-value <0.10) among riparian land uses.
[5]Negative numbers indicate deposition.

TABLE 3. Streambank Erosion Activities Under Different Riparian Land-Uses in Three Iowa Regions.[1]

Riparian Land-Use	Streambank Erosion Activities[2]									
	Year 1[3] (mm)	SD[4]	Year 2[3] (Mm)	SD[4]	Year 3[3] (Mm)	SD[4]	Sum Year 1-3 mm	SD[4]	Average Year 1-3 (mm/year)	SD[4]
Central										
Row-cropped fields	307 (65)	a	298 (59)	a	322 (49)	a	906 (138)	a	302 (46)	a
Continuous pastures	137 (62)	ab	235 (64)	a	336 (52)	a	698 (135)	ab	233 (45)	ab
Rotational pastures	102 (64)	b	137 (74)	ab	252 (58)	ab	491 (137)	bc	164 (46)	bc
Grass filters	165 (65)	b	157 (59)	ab	224 (49)	ab	526 (138)	bc	175 (45)	bc
Riparian forest buffers	107 (65)	b	63 (59)	b	135 (49)	b	285 (138)	c	95 (46)	c
Northeast										
Continuous pastures	182 (55)	a	237 (48)	a	248 (60)	a	679 (110)	a	226 (37)	a
Intensive rotational pastures	141 (65)	a	156 (53)	ab	381 (51)	b	703 (132)	a	234 (44)	a
Cattle fenced out of streams	48 (64)	a	92 (65)	ab	125 (62)	c	295 (139)	b	98 (46)	b
Riparian forest buffers	56 (64)	a	73 (65)	b	105 (62)	c	265 (139)	b	88 (46)	b
Southeast										
Continuous pastures	212 (53)	a	117 (49)	a	247 (52)	a	539 (126)	a	180 (42)	a
Rotational pastures	223 (63)	a	86 (59)	a	250 (52)	a	541 (137)	a	180 (46)	a
Intensive rotational pastures	115 (63)	a	119 (61)	a	265 (49)	a	505 (136)	a	168 (45)	a
Cattle fenced out of stream	102 (90)	a	105 (102)	a	210 (72)	a	459 (211)	a	153 (70)	a
Grass filters	155 (63)	a	82 (61)	a	136 (60)	a	342 (144)	a	114 (48)	a

[1]The mean rainfall that each riparian land-use reach received was used as a covariate to estimate streambank erosion rate. In parentheses is the standard error.
[2]To estimate *erosion activity*, the absolute value of the subtraction between the previous and the most recent measurement of the exposed erosion pin length was used. Streambank erosion activity was the average activity of all the pin plots in the riparian land-use of a region.
[3]Year 1: August 2001-2002; Year 2: August 2002-2003; Year 3: August 2003-2004.
[4]SD, significant differences. In this column the different letters indicate significant differences (p-value <0.10) among riparian land uses.

banks of CP and IP had low erosion rates. Because of this, the FP banks had significantly lower erosion rates than the CP banks. Deposition was probably experienced because of frequent freeze-thaw activities during the winter period and the low streamflows that were not able to remove the deposited material that fell from the top to the bottom of the streambanks during that year. In Year 2 all land-uses experienced erosion, even though precipitation total amounts were lower than in Year 1. During Year 2, the erosion rates and activities on the CP banks were significantly higher than those on the RF and FP banks. In Year 3, CP and IP banks had significantly higher erosion rates and activities than the RF and FP banks. In this year IP banks also had significantly higher erosion rates and activities than the CP banks. This was something we did not expect, although we must note that over the three-year period CP and IP had very similar erosion rates and activities.

Both erosion rate and activity were used to compare among riparian land-uses because they each provide a different perspective on streambank erosion. Erosion rate, measures only the erosional soil bank loss. Pins at the bottom of the bank that have experienced deposition have negative erosion rates. As a result, it can provide a better measure of how much soil was lost to the channel from the streambank. Erosion activity usually results in a larger value because it includes both erosional soil bank loss and depositional soil bank gain. It measures any soil that has moved along the streambank such as soil eroded from the top of the streambank that was deposited at the bottom of the streambank. Some of the depositional soil may only be held temporarily until a large enough discharge event scours it away from the streambank. As a result, erosion activity might measure the loss of the soil twice and overestimate the actual soil lost from the streambank. However, if bank stability is the main interest, erosion activity is probably a better indicator. This is particularly true during dry years, when many streambanks experience primarily deposition of eroded materials (Couper and Maddock, 2001).

Over the three-year period, certain riparian land-uses had higher erosion activities than erosion rates. Specifically, in the central region, the erosion activities of the RF banks were approximately two times greater than erosion rates, while in the northeast region erosion activity for the RF banks was approximately six times greater than erosion rate while erosion activity of the FP banks were approximately 4.5 times greater than erosion rates. Finally, in the southeast region, the erosion activities of GF and FP banks were approximately three times greater than the erosion rates. For all other riparian land-uses in all regions the erosion activities were approximately 1.5 times greater than the erosion rates. The larger differences between erosion activities and erosion rates on the RF, GF, and FP streambanks show that more deposition was occurring on their streambanks. This suggests that RF, GF, and FP streambanks might be stabilizing faster than those in the other riparian land-uses, especially if plants are able to colonize the depositional material at the bottom of the banks. The banks of RF, GF, and FP are transitioning faster from the widening phase (Stage III) to the stabilizing phase (Stage IV) of the channel evolution model (Schumm *et al.*, 1984) because of fewer disturbances and plant establishment.

The maximum pin plot erosion rate ranked as follows: in the central region, RC > GF > CP > RP > RF; in the northeast region, CP > IP > RF > GF; and in the southeast region, FP > CP > RP > IP > GF. The maximum pin plot erosion rates did not follow the same order as the erosion rates and activities. In this study these extremes were as much as 3.5 times greater than the mean erosion rates for certain riparian land-uses in the central region, as much 5.3 times greater in the southeast region and as much as 9.5 times greater in the northeast region (Tables 2 and 4). Using only the mean erosion rates can greatly underestimate how much a specific streambank can potentially retreat after a significant erosion event. Interestingly, the highest extremes between the maximum pin plot rates and the average erosion rate were found in land-uses that excluded livestock from the channels. Banks of the RF (3.5 times) and GF (3.0 times) in the central region, banks of RF (9.5 times) and FP (4.2 times) in the northeast region, and banks of FP (5.2 times) in the southeast region had the largest differences between mean erosion rates and the maximum pin plot erosion rates. This might indicate that most banks along the RF, GF, and FP are stabilizing but that they have a few outside bend banks that are still highly erosive. Streambank erosion is a natural process and streams should be expected to have some banks that are highly erosive.

In most cases erosion rates and activities were lower, although not always significantly different, on banks of RF, GF, and FP than on those along crop fields or pastures. The RF, GF, and FP had perennial plant communities along their banks and no livestock pressure. In many cases, erosion rates and activities were between two and five times lower. The RC banks had the highest erosion rates and activities followed by the various grazing practices. Grazing practices that allowed direct access to the stream channel were not significantly different from each other regardless of the amount of rest that was allowed for plant regrowth in the paddocks. Bank healing seems

TABLE 4. Streambank Maximum Pin Plot Erosion Rates Under Different Riparian Land-Uses in Three Iowa Regions.[1]

	Streambank Maximum Pin Plot Erosion Rate[2]				
Riparian Land-Use	**Year 1[3] (mm)**	**Year 2[3] (mm)**	**Year 3[3] (mm)**	**Sum Year 1-3 (mm)**	**Average Year 1-3 (mm/year)**
Central					
Row-cropped fields	551	569	586	1246	415
Continuous pastures	309	300	442	865	288
Rotational pastures	184	345	319	639	213
Grass filters	195	417	450	954	318
Riparian forest buffers	70	53	406	479	160
Northeast					
Continuous pastures	461	514	512	1487	496
Intensive rotational pastures	883	367	511	1386	462
Cattle fenced out of streams	36	51	292	285	95
Riparian forest buffers	178	10	276	423	141
Southeast					
Continuous pastures	387	96	497	834	278
Rotational pastures	387	57	347	735	245
Intensive rotational pastures	272	206	372	683	228
Cattle fenced out of stream	249	313	348	910	303
Grass filters	128	16	147	171	57

[1]The mean rainfall that each riparian land-use reach received was used as a covariate to estimate streambank erosion rate.
[2]*Maximum pin plot erosion rate* is the erosion rate of the pin plot with the highest erosion rate within each riparian land-use in each region.
[3]Year 1: August 2001-2002; Year 2: August 2002-2003; Year 3: August 2003-2004.

to require more time than regrowth of the forage in the paddocks.

Severely Eroding Bank Survey

Lyons *et al.* (2000) found that 1-66% of the streambank lengths of streams surveyed in Wisconsin were severely eroding, similar to the 10-54% found in this study. The lowest percentage (10%) was found along RF reaches in northeast Iowa while the highest (54%) was found along the CP and RP reaches in the southeast (Table 5). In many cases riparian land-use practices that had perennial vegetation and excluded livestock (RF, GF, and FP) had significantly lower percentages than the riparian agricultural land-uses (RC, CP, RP, and IP) (Table 5). Among the grazing riparian land-uses the only significant difference was found in the southeast, where IP was lower than the other two grazing practices. In Wisconsin, Lyons *et al.* (2000) found significantly higher percentages of severely eroding banks in CP reaches than in those of IP, GF, and RF. In southwestern Wisconsin, Simonson *et al.* (1994) suggested that streams of high quality should have less than 20% of their streambank lengths severely eroding. In this study, severely eroding streambank lengths along RF, GF, and FP in all regions were always below this percentage. In contrast, the CP, RP, IP, and RC had 25% or more of their streambank lengths severely eroding across all regions.

The percentage of severely eroding bank lengths showed more significant differences among riparian land-uses (Table 5) than the erosion rate or activity (Tables 2 and 3). As the erosion pins were placed on severely eroding banks, high erosion rates and activities were expected. The data from this study suggest that the percentage of severely eroding bank lengths provides a better indicator of the impacts of the adjacent riparian areas.

Soil and Phosphorus Losses

Total soil and phosphorus losses among riparian land-uses were strongly correlated to the lengths of severely eroding streambanks (Table 5). Once again, RF, GF, and FP streambanks had the lowest losses regardless of region. For these land-uses, soil and phosphorus losses were in the range of 2-48 times and 2-62 times less, respectively than the agricultural land-uses (RC, CP, RP, and IP). Among the grazing practices, IP streambanks had the lowest losses with no consistent differences between CP and RP streambanks.

Streambanks along RC in the central region had the highest soil losses with 304 tonne/km/year (Table 5). Across all regions, streambank soil losses along CP ranged from 197 to 264 tonne/km/year, while those along RP ranged from 94 to 266 tonne/km/year, and those along IP ranged from 124 to 153 tonne/km/year. Streambanks along FP and GF

TABLE 5. Soil and Total Phosphorus Losses From Streambank Erosion Under Different Riparian Land-Uses in Three Iowa Regions.[1]

Riparian Land-Use	Stream Reach Length	Severely Eroding Streambank			Bulk Density[2] (tonne/m³)	Streambank Soil Loss[3]		Streambank Soil Phosphorus Concentrations[2] (kg/tonne)	Streambank Phosphorus Loss[4]	
	Total (km)	Length %	SD[5]	Area (m²)		Total (tonne/year)	Unit length (tonne/km/year)		Total (kg/year)	Unit Length (kg/km/year)
Central										
Row-cropped fields	1.6	44 (6)	a	1657	1.23	487	304	354	172	108
Continuous pastures	1.7	39 (6)	ab	1999	1.35	448	264	349	156	92
Rotational pastures	1.3	25 (6)	b	899	1.31	122	94	398	49	37
Grass filters	1.6	16 (6)	bc	615	1.16	76	47	303	23	14
Riparian forest buffers	1.4	14 (6)	bc	430	1.24	25	18	350	9	6
Northeast										
Continuous pastures	1.6	38 (5)	a	1935	1.15	381	238	518	197	123
Intensive rotational pastures	1.5	27 (5)	a	1125	1.20	230	153	432	99	66
Cattle fenced out of streams	0.8	11 (6)	b	203	1.16	5	6	464	2	3
Riparian forest buffers	0.8	10 (6)	b	244	1.10	4	5	479	2	2
Southeast										
Continuous pastures	1.8	54 (5)	a	2661	1.32	355	197	360	128	71
Rotational pastures	1.5	54 (6)	a	2403	1.36	399	266	459	183	122
Intensive rotational pastures	0.7	32 (6)	b	371	1.28	87	124	531	46	66
Cattle fenced out of stream	0.3	16 (6)	bc	239	1.32	18	61	555	10	34
Grass filters	0.7	16 (6)	c	289	1.29	15	22	406	6	9

[1]The mean rainfall that each riparian land-use reach received was used as a covariate to estimate streambank erosion rate. In parentheses is the standard error.
[2]Data from Zaimes *et al.*, 2008.
[3]The product of streambank erosion rate, bulk density, and severely eroding bank area for each riparian land-use within a region was used to estimate its *total soil loss* from the streambanks. Streambank erosion rate was the average rate of all the pin plots in the riparian land-use of a region. Streambank s*oil loss per unit of stream length* was estimated by dividing the streambank phosphorus loss for each riparian land-use by its total stream reach length within a region.
[4]The product of streambank erosion rate, bulk density, severely eroding bank area, and average soil phosphorus concentration for each riparian land-use within a region, was used to estimate its *total phosphorus loss* from the streambanks. Streambank erosion rate was the average rate of all the pin plots in the riparian land-use of a region. Streambank *phosphorus loss per unit of stream length* was estimated by dividing the streambank soil loss for each riparian land-use by its total stream reach length within a region.
[5]SD, significant differences. In this column the different letters indicate significant differences (p-value <0.10) among riparian land uses.

had soil loses that ranged from 6 to 61 tonne/km/year and 22-47 tonne/km/year respectively, while those along RF had losses ranging 5-18 tonne/km/year. In Vermont, DeWolfe *et al.* (2004) found similar soil losses (10-663 tonne/km/year) from streams with similar watershed areas to this study.

Total phosphorus concentration differences in streambank soils among riparian land-uses (Table 5) were not significant (Zaimes *et al.*, 2008). Total phosphorus losses from streambanks along RC in central Iowa were 108 kg/km/year. Across all regions total phosphorus losses along CP ranged from 71 to 123 kg/km/year, while along RP losses ranged from 37 to 122 kg/km/year, and along IP losses were 66 kg/km/year. Banks along FP, GF, and RF had the smallest phosphorus losses ranging from 3 to 34, 9-14, and 2-6 kg/km/year, respectively. Large phosphorus losses per unit length from banks along pastures with full livestock access to the stream and row-cropped fields indicate that streambank erosion can be a significant contributor to the stream water phosphorus load. Similar streambank phosphorus losses (10-840 kg/km/year) from streams with similar watershed areas to this study were found in Vermont (DeWolfe *et al.*, 2004).

Streambank Erosion and Riparian Land-Uses

Based on the responses of the six variables (Tables 2, 3, 4, and 5) used in this study, RF was the land-use that stabilized streambanks and minimized soil and total phosphorus losses the most. These responses are especially encouraging because most of the RF had only recently been established following the abandonment of past riparian management practices such as RC and CP. The GF riparian land-use followed but was not as efficient. This could have been because the GF were even younger than the RF in some cases. In addition, tree root systems probably provide more protection to streambanks than grass roots along the deeply incised channels with nearly vertical banks that were found along our study reaches. There has been a lot of debate about the role of roots in bank stabilization, with some indicating tree roots as more effective (Gregory *et al.*, 1991), while others suggest grass roots are more effective (Lyons *et al.*, 2000). Recent studies indicate that trees stabilize streambanks better because of the greater quantity of larger diameter roots (Wynn *et al.*, 2004; Wynn and Mostaghimi, 2006). In general, when selecting riparian vegetation for streambank stability, it is very important to not only consider the hydrologic channel processes but also the mechanical and ecological processes that control streambank stability (Simon and Collison, 2002).

Regardless of whether it is trees or grasses, perennial plant communities with vigorous root systems increase streambank stability. Vegetation is an integral part of the riparian landscape, and the amount of streambank vegetation, especially in low-order streams, is important because of the stabilizing support the roots can provide (Thorne and Tovey, 1981). However, when bank height exceeds the rooting depth of the vegetation other stream stabilization techniques might be necessary.

In FP reaches, streambank stability was greater than in the other grazing systems that allowed full cattle access to the stream. Cattle are attracted to riparian areas and tend to spend a lot of time in and around the stream (Trimble and Mendel, 1995). Improvements in streambank stability in FP reaches have also been found in other studies (Laubel *et al.*, 2003), but this practice is not socially and economically acceptable to many farmers in Iowa. Where off-stream water is provided as an alternative to fencing, streambank erosion has been dramatically reduced (Sheffield *et al.*, 1997), and in some cases cattle weight gains have even been seen (Porath *et al.*, 2002). In Iowa, off-stream water without fencing would not be as effective as it is in some other states because many pastures are confined to the narrow riparian corridors along low-order streams.

There were mixed results when comparing RP and IP to CP. In most cases CP had the greatest negative impact on streambanks. There were indications, although mostly nonsignificant, that IP had less impact on streambanks than either RP or CP. The differences in individual farmer interpretation of each of these practices and the fact that some of the RP and IP systems had been established for no longer than three years may have contributed to the inconsistencies that were found in this study. Work by Lyons *et al.* (2000) suggested that IP can improve streambank stability and decrease soil losses. Decreased erosion and increased stability could be attributed to the shorter time cattle spend in the stream and the adjacent riparian areas thereby reducing streambank disturbance. Bank stabilization could probably increase more if the number of paddocks along the stream decreased and the number of paddocks in the uplands increased. This would decrease the time that cattle spent in the riparian areas. Even with decreased numbers of paddocks there may not be enough rest to allow plants to get reestablished on heavily disturbed streambanks. So in many cases, the keys to successful recovery of streambank stability in pastures will include decreasing animal stocking rates, controlling the timing of grazing in the riparian paddocks, especially under wet conditions and when the least damage to the plants can be done (Clary and Kinney, 2002), or by eliminating cattle from the streambanks completely until plants are re-established.

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