FINAL PROJECT REPORT

ECOLOGICAL EFFECTS OF STORMWATER AND STORMWATER CONTROLS ON SMALL STREAMS

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LINKAGES BETWEEN WATERSHED AND STREAM ECOSYSTEM CONDITIONS IN THREE REGIONS OF THE UNITED STATES

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EXECUTIVE SUMMARY

RELATIONSHIPS AMONG WATERSHED CONDITIONS AND STREAM BIOLOGY

Defining Watershed and Biological Conditions

Coordinated studies in three different regions in the United States related stream biological communities to land use and land cover attributes of the watersheds draining to their habitats. Biological communities were defined in terms of multi-metric benthic macroinvertebrate indices developed for each region and indices characterizing fish communities in two locations. The Benthic Index of Biotic Integrity (B-IBI) characterized invertebrate communities in the Puget Sound Lowlands, Washington, and Montgomery County, Maryland. A similar formulation was developed for the region of Austin, Texas. A Fish Index of Biotic Integrity (F-IBI) represented the Montgomery County streams. Puget Sound fish were studied using the ratio of the presence of a species relatively intolerant to stress (coho salmon) to a more tolerant fish (cutthroat trout; young-of-the-year in both cases).

Geographic information system (GIS) analysis delineated watershed pervious and impervious cover. GIS data were used to develop multi-metric Watershed Condition Indices (WCIs) for each region. Metrics comprising the WCIs are either relatively highly correlated to biological indices or were identified in preliminary stepwise multiple and logistic regression exercises as instrumental in linking watershed and aquatic biological states. Among all three regions the landscape variables most associated with biological metrics are measures of total impervious area and forest cover at the watershed scale, in riparian buffer zones over a range of widths, and in local areas not necessarily in riparian zones but within certain distances near streams.

General Observations

The highest biological indices in all cases are associated only with the highest WCI values, representing no or extremely low urban development, very high forest retention, and minimal human intrusion in riparian zones. It was therefore demonstrated in three different regions of the nation that the best biological health is impossible unless human presence is very low <u>and</u> the natural vegetation and soil systems are well preserved near streams and throughout watersheds. However, while these conditions are necessary for high integrity, they are not sufficient by themselves to guarantee it. Other circumstances not captured in the GIS-based watershed analysis must also be instrumental.

An additional observation common among regions was that biological responses to urbanization in combination with loss of natural cover do not exhibit thresholds of watershed change that can be absorbed with little decline in health. Instead, decline was seen to start in the earliest stages of land conversion to human occupation. Furthermore, in all three regions comparatively high urbanization and natural cover loss make relatively poor biological health the inevitable outcome.

WCI in the range of 70 to 80 percent of the best watershed condition is essential in all three regions to attain at least 80 percent of best macroinvertebrate community integrity. A watershed index at least at the lower end of this range is also necessary for clear dominance (in the ratio of at least 3:1) of Puget Sound coho salmon instead of the more tolerant cutthroat trout. At the opposite end of the biological spectrum, invertebrate community health under 40 percent of best integrity occurs in each region, excepting only two cases, at WCI = 25-30 percent of best condition and below. Cutthroat trout dominance is also assured under these watershed conditions in the Puget Sound streams.

Quantification of Results

Several statistical and multivariate analytical techniques were applied to devise formal mathematical constructs to increase the utility of the biological and watershed indices as assessment and management tools. Logistic regression and discriminant function analyses proved to be useful mechanisms for classifying biological integrity according to watershed condition.

Logistic regression analysis produced equations forecasting the probability of a stream's invertebrate or fish community being in selected groupings of biological integrity based on WCI. The equations were more successful in predicting that a site would not have a certain biological condition than forecasting that it would fall in the specified group. Hardest to forecast with these models is very good biological community health. This consistent observation across regions is another reflection of the necessity but not sufficiency of relatively high WCI for high biological integrity. The models were more successful, although still inconsistent, in predicting membership in degraded biological groups than in high quality categories. The greater success in forecasting exclusion from rather than inclusion within a group makes logistic regression best suited to analyze if it is possible, with the existing or expected watershed condition, for a stream to achieve a high level of biological integrity or avoid a low level.

Discriminant function analysis generates a new variable (the discriminant function) from the independent variable(s) and uses it with a statistical criterion to classify values of the independent variable in groups. This technique was more successful than logistic regression in judging actual membership in relatively high and low biological integrity groups. The two methods can be used in concert to assist in judging how likely a certain biological state is for a particular case. It must always be recalled, though, that actual achievement of the best biological health depends on some factors yet to be defined.

For illustration, the Puget Sound logistic regression equations were applied to hypothetical watershed conditions. The results give strong evidence of very low probability for relatively healthy invertebrate and fish communities with WCI much under 70 percent. WCI in the range from 79 down to 57 percent of best condition is a region of rapid loss of prospects for high biological integrity. B-IBI ≤ 45 percent of best integrity

is highly probable as WCI goes below 45 percent. Decline of the coho salmon:cutthroat trout ratio to 1.0 is very likely around the same WCI. A heavily depleted benthic community (B-IBI ≤ 25 percent) becomes probable just under WCI of 20 percent.

THE ROLE OF STRUCTURAL STORMWATER BEST MANAGEMENT PRACTICES IN STREAM BIOLOGICAL INTEGRITY

Extensive and incisive investigation of how stormwater BMPs affect the portrait of aquatic biology in relation to overall watershed conditions was hindered by the very labor-intensive effort required to collect meaningful data on the numerous BMPs that often exist in urban watersheds. With this dilemma the study proceeded in two directions: (1) a broad approach over all watersheds with recorded BMP presence to determine if the mere extent of BMP coverage, with no assessment of implementation quality, has an identifiable, positive effect on stream health; and (2) a deeper effort in a few watersheds to collect and evaluate the data necessary to gauge BMP implementation quality and its effect on aquatic systems.

The broad-scale approach was not very fruitful. Early graphical plots of biological versus urbanization measures for catchments with and without BMPs did not distinguish differences in biological quality between the two groups. Follow-up statistical examinations of BMP areal coverage, with overall watershed condition being a controlled variable, exhibited very weak or even negative partial correlations between biological integrity and BMP presence.

In the second, deeper approach, structural BMPs were intensively studied in several subbasins of two Puget Sound stream systems, one with perhaps the greatest consideration to stormwater management in the region and the other with less attention. Even in the first watershed, a minority of the developed area is served by runoff quantity control practices, and even less of it by water quality control BMPS. Those BMPs installed are capable of mitigating an even smaller share of urban impacts, primarily because of inadequacies in design standards.

Even with these shortcomings, though, results indicate that structural BMPs appear to help in sustaining aquatic biological communities at fairly high urbanization levels. They give less evidence of benefit at moderate urbanization and greater natural land cover. If ecological losses are to be stemmed at high urbanization, structural BMPs appear to have a substantial role.

The highest biological indices had no relationship to BMPs, because these high scores occurred only in watersheds with no or minimal development, where no BMPs were built. It thus could not be tested if BMPs can replace some loss in natural land cover through light urbanization and still maintain high biological integrity. However, the lack of obvious benefit seen with a moderate amount of development lends support to the hypothesis that the benefit would also be absent at low urbanization too, where relatively undisturbed streams house the most sensitive organisms.

RECOMMENDATIONS

Unity in the results from three dispersed and differing areas of the nation support certain general watershed management recommendations for strong consideration elsewhere. This work also developed methods that can be broadly recommended to assist any region wishing to develop a basis for its own watershed analysis and management efforts.

General Watershed Management Recommendations

1. Base watershed management on specific objectives tied to desired biological outcomes.

2. If the objective is to attain an existing high level of biological integrity, very broadly preserve the extensive watershed and riparian natural vegetation and soil cover almost certainly present through mechanisms like outright purchase, conservation easements, transfer of development rights, etc.

3. If the objective is to prevent further degradation when partially developed areas urbanize further, maximize protection of existing natural vegetation and soil cover in areas closest to the stream, especially in the nearest riparian band. In the uplands, generally develop in locations already missing characteristic natural vegetation. As much as possible, preserve existing natural cover and limit conversion to impervious surfaces. The lower the level of existing development, the more important this recommendation is.

4. In addition, fully serve newly developing and redeveloping areas with stormwater quantity and quality control BMPs sited, designed, and operated at state-of-the-art levels. Attempt to retrofit these BMPs in existing developments. The higher the level of existing development, the more important this recommendation is; since much opportunity to apply the preceding recommendation is lost with extensive land conversion.

5. Where riparian areas have been degraded by encroachment, crossings, or loss of mature, natural vegetation, give high priority to restoring them to extensive, unbroken, well-vegetated zones. This strategy could be the most effective, as well as the easiest, step toward improving degraded stream habitat and biology. Riparian areas are more likely to be free of structures than upland areas and more directly influence stream ecology. Also, riparian restoration fits well with other objectives, like flood protection and provision of wildlife corridors and open space

6. The above recommendations suggest that federal and state environmental management agencies should reconsider their existing water body classification systems and the associated water quality standards. This is consistent with the recommendation of the National Academy of Sciences (NRC 2001) review of the nation's total maximum daily load (TMDL) program that states needed to conduct

use attainability analyses and appropriate designate the beneficial uses of water bodies. State watershed managers need to work closely with local communities to develop water body classifications that accurately reflect the desired and achievable goals of the community for its aquatic ecological systems.

Recommendations for Developing Regional Watershed Analysis and Management Approaches

1. Systematically collect data on regionally representative stream benthic macroinvertebrate and fish communities. Extend the program's coverage over the full range of urbanization, from none to the highest levels with aboveground streams. Use the data to develop regionally appropriate biological community indices.

2. Develop a geographic information system to organize and analyze watershed land use and land cover (LULC) data. Collect data on regionally appropriate LULC variables, particularly measures of impervious and forested cover in the watershed as a whole, at least two riparian bands extending to points relatively near and far from the stream, and in other local areas fairly close to the stream.

3. Investigate which LULC variables are statistically best associated with biological indices, using analyses like correlation and stepwise multiple and logistic regressions.

4. Define a tentative Watershed Condition Index (WCI) using the best-associated variables.

5. Choose the optimum LULC variables for the WCI on the basis of the combination yielding the best fits in statistical regressions of biological indices on WCI. These regressions are useful for fine-tuning the WCI but are unlikely to offer very good tools for predicting biology as a function of WCI.

6. Validate the resulting WCI with discriminant function analyses as described in this report.

7. Graph biological indices versus WCI and examine trends signifying potentially fruitful regional watershed management strategies.

8. Perform logistic regression analyses to develop means of classifying probable groupings of aquatic biological health in relation to WCI. This type of analysis was found in this study to be better at predicting if a particular case would not be in a group than if it would be.

9. Supplement the logistic regressions with discriminant function analysis, which was found to be better at forecasting if a case would fall in a group.

10. Use the two techniques in concert to make judgments like: (1) With prevailing or expected watershed conditions, is it possible for a biological state to be at the highest level or, in other situations, avoid the lowest level? (2) With these conditions, how likely is it that the state will actually attain that level? (3) What management strategies can be considered, and are most likely to be feasible and successful, to adjust watershed conditions in a way that will maximize the chance of attaining a biological objective?

Deliverables from this project

The following products were developed during this multiple-phased project:

- Montgomery County, Maryland Project, The Ecological Response of Small Streams to Stormwater and Stormwater Controls. 1998. Montgomery County Dept. of Environmental Protection and the Watershed Management Institute.
- City of Austin, Texas Project, The Ecological Response of Small Streams to Stormwater and Stormwater Controls. 1998. Austin Watershed Protection Department and the Watershed Management Institute.
- Vail, Colorado Project, The Ecological Response of Small Streams to Stormwater and Stormwater Controls. 1998. Town of Vail Community Development Department and the Watershed Management Institute.
- Puget Sound, Washington Project, The Ecological Response of Small Streams to Stormwater and Stormwater Controls. 1998. Dr. Richard R. Horner and Dr. Christopher W. May and the Watershed Management Institute.
- Ecological Effects of Stormwater and Stormwater BMPs on Small Streams: Lessons Learned and Recommendations to Improve Quality Control in Macroinvertebrate Bioassessments. 2003. John Maxted and Eric H. Livingston, Watershed Management Institute
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- May, C.W. and R.R. Horner. 2002. The Limitations of Mitigation-Based Stormwater Management in the Pacific Northwest and the Potential of a Conservation Strategy Based on Low-Impact Development Principles. Proc. 2002 ASCE Stormwater Conference, Portland, OR.
- Blaha, D.W., C. W. May, and R. R. Horner. 2002. The effectiveness of stormwater management and riparian buffers in mitigating the effects of urbanization on streams. Proceedings of the Watershed 2002 Conference, February 2002, Water Environment Federation, Alexandria, Virginia.

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TABLE OF CONTENTS

Page

LIST OF FIGURES

LIST OF TABLES

BACKGROUND

WATER RESOUCE MANAGEMENT EVOLUTION

The foundation for protecting and managing the nation's water resources is the Federal Water Pollution Control Act Amendments of 1972 (PL 92-500) as amended by the Clean Water Act of 1977 and the Water Quality Act of 1987. Since its enactment, this law has guided the implementation of programs, regulations, and activities by federal, state and local governments, and by the private sector to achieve the act's goals of restoring and maintaining the chemical, physical, and biological integrity of the Nation's waters. During this 30 year period, an evolution has occurred as the focus on our nation's water resources has moved from the control of point sources, to the control of nonpoint sources, to the management of the cumulative effects of human activities.

In the beginning, the focus of water pollution control programs was on reducing the discharge of pollutants from traditional point sources such as domestic and industrial wastewater discharges. The 1972 amendments led to the establishment of nationally established effluent limitations that could be specified in National Pollutant Discharge Elimination System (NPDES) permits. The effluent limitations were based on best available technologies appropriate for the specific type of wastewater. In some water bodies, water quality based effluent limits that required even higher levels of treatment were established if needed to meet local water quality goals. As a result of these actions, pollutants discharged by industrial and municipal wastewater plants declined and the water quality of many receiving waters improved.

In the late 1970s, as the Section 208 program was being implemented, the focus on water pollution shifted to nonpoint sources such as urban and agricultural runoff, construction site runoff, and leaching from farm fields or onsite wastewater systems. The results of numerous Section 208 monitoring studies provided data demonstrating how significant nonpoint source pollutant dischargers were to the degradation of our nation's surface and ground waters. Unfortunately, very few states implemented comprehensive nonpoint source management programs in response to the Section 208 program. However, a few states and enlightened regional or local governments did begin implementing programs that required the treatment of stormwater from new urban developments. These efforts provided essential information on the ability and costs of best management practices (BMPs) to reduce stormwater pollution. BMPs are control techniques used in a given set of conditions to provide water quantity and water quality enhancement in the most cost effective manner.

The Water Quality Act of 1987 brought further focus on "wet weather" discharges such as stormwater runoff and other nonpoint sources of pollution. Under section 402(p) of the Water Quality Act, certain urban stormwater discharges were reclassified as point sources of pollution and were now subject to regulation under NPDES permits. Additionally, Section 319 of the Act required states to assess the impact of nonpoint source pollution and to develop programs to minimize their impact on surface and ground waters. Furthermore, Congress appropriated significant funding (currently

around $240 million per year) to states to be used for implementing nonpoint source management programs. These funds also can be used to demonstrate and evaluate the effectiveness of various BMPs in reducing stormwater and other types of nonpoint source pollution.

The implementation of "technology-based controls" for wastewater and stormwater discharges has helped improve water quality of our nation's rivers, lakes, and estuaries. However, this approach also has had unintended consequences (NAS, 2001) including:

- focus on compliance with permit conditions, not the health of aquatic ecosystems;
- focus on chemical pollutants and water chemistry dominated water quality policy;
- lack of focus on the biological and physical determinants of water body health;
- lack of focus on understanding the complex interactions within a watershed that ultimately determine the health of aquatic ecosystems.

The most recent evolution of water quality management in the United States has been the implementation of the total maximum daily load (TMDL) program required by Section 303(d) of the Federal Clean Water Act. The TMDL program and the growing emphasis by EPA and states on watershed management is focusing water policy on the restoration or enhancement of the health of water bodies whose beneficial uses have been adversely impacted by the cumulative effects of human activities. Managing the cumulative effects of human activities throughout a watershed demands a much higher level of scientific knowledge, especially of the linkages between watershed characteristics (hydrologic, ecological, anthropogenic) and aquatic ecosystem health.

URBAN STREAMS AND THEIR MANAGEMENT

By the mid-point of the 1990s the effects of watershed urbanization on streams were well documented. They include extensive changes in basin hydrologic regime, channel morphology, and physicochemical water quality associated with modified rainfall-runoff patterns and anthropogenic sources of water pollutants. The cumulative effects of these alterations produce an in-stream habitat considerably different from that in which native fauna evolved. In addition, development pressure has a negative impact on riparian forests and wetlands, which are intimately involved in stream ecosystem functioning. Much evidence of these effects exists from studies of urban streams around the United States (e.g., Klein 1979; Richey 1982; Pedersen and Perkins 1986; Scott, Steward, and Stober 1986; Garie and McIntosh 1986; Booth 1990, 1991; Limburg and Schmidt 1990; Booth and Reinelt 1993; Weaver and Garmen 1994).

What was missing at that point in time, though, was definition of the linkages tying together landscapes and aquatic habitats and their inhabitants strong enough to support management decision-making that avoids or minimizes resource losses. Lacking this systematic picture, urban watershed and stormwater management efforts have not been broadly successful in fulfilling the federal Clean Water Act's stipulation to protect the biological integrity of the nation's waters. Effective management needs:

- well-conceived goals of what biological organisms and communities are to be sustained and at what levels;
- understanding the foundation for judging what habitat conditions they need for sustenance; and, in turn
- understanding the watershed attributes consistent and inconsistent with these habitat conditions.

To date, management has usually centered on attempting to reduce stormwater runoff contaminants in passive structural BMPs like wet detention ponds with permanent pools or extended detention, vegetated conveyance such as swales, infiltration basins, sand filters, and others. Numerous projects around the country have monitored the effectiveness of these BMPs in reducing stormwater pollutant loadings or concentrations. Some stormwater programs also focused management attention on amelioration of stormwater peak flow rate increases following development to reduce stream erosive shear stress and its damage to stream habitats. However, there has been little tie between these prescriptions and ecological considerations, or even how well they work to sustain biological communities that they ostensibly exist to protect.

What little study had been done was far too limited to draw firm conclusions but was not promising. Maxted and Shaver (1997) were not able to distinguish a biological advantage associated with the presence of structural BMPs serving eight Delaware stream reaches versus their absence in 33 cases. Jones, Via-Norton, and Morgan (1997) studied biological and habitat response in streams receiving discharges from several types of water quality and quantity control BMPs relative to reference locations. They concluded that appropriately sited and designed BMPs provided some mitigation of stormwater impacts, but that the resulting communities were still greatly altered from those in undeveloped watersheds.

TOWARD A MORE SYSTEMATIC VIEW OF WATERSHEDS, STREAMS, AND MANAGEMENT

With this background of insufficient understanding of relationships among watershed and aquatic ecosystem elements, and the capabilities of prevailing management strategies to influence these relationships, the U.S. Environmental Protection Agency (USEPA) commissioned the Watershed Management Institute (WMI) to investigate "The Ecological Effects on Small Streams of Stormwater and Stormwater Controls." This project would study stream habitats and biology across gradients of urbanization and BMP application in four regions of the nation (Austin, TX; Montgomery County, MD; Puget Sound, WA; and Vail, CO). The hypothesis being tested was that the implementation of structural BMPs to reduce pollutant loadings and peak discharge rates would allow higher levels of biological integrity to occur at higher levels of watershed imperviousness. Additionally, the project would examine the effectiveness of nonstructural controls in reducing urban stormwater impacts on small streams. Initital results of the project when combined with other data from the Puget Sound region led to the later phases of the project looking more closely at the role of nonstructural BMPs in protecting stream biological conditions.

Phase 1 of "The Ecological Effects on Small Streams of Stormwater and Stormwater Controls" was conducted in Montgomery County, Maryland and Austin, Texas. Phase 2 of the project was conducted in Vail, Colorado where the focus was primarily on nonstructural BMPs, and in the Puget Sound region of Washington. Phase 3 of the project led to collecting additional watershed characteristic information in Montgomery County, Austin, and Vail, especially on stream riparian zones, wetland retention, and forest retention. Phase 3 work also focused on two Puget Sound subwatersheds where detailed information on stormwater BMPs was collected. Finally, Phase 4 of the project built on some of the insights gained from the earlier phases to refine the relationships between structural and nonstructural BMP implementation and stream ecosystem health.

This study followed an earlier effort along similar lines in the Puget Sound region funded by the Washington Department of Ecology Centennial Clean Water Fund. Together these studies built a database now totaling more than 650 reaches on low-order streams in watersheds ranging from no urbanization and relatively little human influence (the reference state, representing "best attainable" conditions) to highly urban (>60 percent total impervious area, TIA).

Results from the initial Puget Sound research and a portion of the follow-up study have been extensively reported. Biological health was assessed according to the benthic index of biotic integrity (B-IBI; Fore, Karr, and Wisseman 1996) and the ratio of young-of-the-year coho salmon (a relatively stress-intolerant fish) to cutthroat trout (a more stress-tolerant species). Both biological measures declined with TIA increase without exhibiting a threshold of effect; i.e., declines accompanied even small levels of urbanization (May 1996; Horner et al. 1997; May et al. 1997). However, stream reaches with relatively intact, wide riparian zones in wetland or forest cover exhibited higher B-IBI values than reaches equivalent in TIA but with less riparian buffering. Until TIA exceeded 40 percent, biological decline was more strongly associated with hydrologic fluctuation than with chemical water and sediment quality decreases. Accompanying hydrologic alteration was a loss of habitat features, like large woody debris and pool cover, and deposition of fine sediments that reduce dissolved oxygen in the bed substrata where salmonid fish deposit their eggs. The research defined stream quality zones in relation to TIA and riparian corridor condition. Most importantly, it identified sets of necessary, although by themselves not sufficient, conditions to maintain a high level of biological functioning or prevent decline to a low level. These findings began to provide a basis for managing watersheds in relation to biological goals.

Maxted (1999) gave a preliminary report on the overall results available at that time of Phases 1 and 2 of the WMI study. Differences in expressions of macroinvertebrate community integrity appropriate for the various locations were reconciled by scoring each relative to the best attainable measure for the region. The patterns of association between these biological expressions and TIA were similar for the Maryland, Texas, and Washington sites, and also similar to the Delaware watersheds studied earlier (Maxted and Shaver 1997). Namely, none exhibited a threshold level of urbanization where biological decline began. As the Delaware results had indicated, stream reaches

studied by WMI with and without structural BMPs could not be distinguished in biological quality. This preliminary analysis pointed out two instances of general unity among differing ecoregions in landscape-aquatic ecosystem relationships – the importance of both riparian and forest retention within a watershed.

Phase 3 of the project returned to the Puget Sound area to focus more on the question of BMP effectiveness. This investigation considered the density of structural BMP coverage and, as *de facto* non-structural BMPs, extent of watershed forest cover and riparian buffering (proportion of upstream corridor with riparian zone in forest or wetland cover at least 30 meters wide on each bank). In this comparison, riparian retention exhibited greater and more flexible potential than other options to uphold biological integrity when development increases. Upland forest retention also offered valuable benefits, especially at lower levels of the urbanization gradient (Horner and May 1999). Structural BMPs at the prevailing densities demonstrated less potential than the non-structural methods assessed to forestall resource decline as urbanization starts and progresses. There was a suggestion in the data, though, that more thorough coverage would offer substantive benefits in this situation. Moreover, structural BMPs were seen to help prevent further resource deterioration in moderately and highly developed watersheds. Analysis showed that none of the options is without limitations, and widespread landscape preservation must be incorporated to retain the most biologically productive aquatic resources.

REFINEMENT OF LINKAGES AMONG WATERSHED CONDITIONS, STRUCTURAL AND NON-STRUCTURAL BMPS, AND STREAM ECOLOGY (Phase 3)

Scope of the Analysis

Observation in the Puget Sound study area of the role played by riparian and upland forest retention in maintaining stream ecology suggested that their benefits might be found in other regions having different aquatic ecosystems. If similarity were demonstrated, the finding would not only serve the pragmatic need for targeting management attention, but would also continue to develop the picture of general unity among ecoregions. In Phase 3 of the USEPA and WMI project, the hypothesis was tested in the Montgomery County, Austin, and Vail study areas using the data collection and analysis methods developed in the Puget Sound study. The next section of the report, Comparison of Ecological Benefits of Riparian and Forest Retention in Four Ecoregions, summarizes the results.

Results from initial Puget Sound work further suggested that structural BMPs can make a substantive contribution to keeping stream ecosystem health from falling to the lowest levels at moderately high urbanization and, with extensive coverage, to maintaining relatively high biotic integrity at light urbanization. It is common sense that service level (coverage) should make a difference, but also that quality of implementation (design, construction, operation, and maintenance) should likewise matter. Also as part of the

next phase, the analysis by Horner and May (1999) was supplemented by more detailed evaluation of service level and added assessment of implementation quality in several catchments relatively well and poorly served with structural BMPs. A later section, Detailed Puget Sound Structural BMP Assessment, summarizes the findings.

These two refinement tasks made up Phase 3 of the USEPA and WMI research project. Horner et al. (2002) presented more complete details of the analyses summarized in the next two report sections.

Comparison of Ecological Benefits of Riparian and Forest Retention in Four Ecoregions

Study Sites and Methods—

The four regional programs had accumulated data from 229 stream reaches by the point of this analysis. Invertebrate data from each program were used to develop multi-metric community indices appropriate for prevailing ecological attributes but similar in complexity (refer to Appendix A). Vail watershed configurations differ substantially from the others, because of topography and other physiographic factors and the development patterns prevalent there. Most Vail area streams originate in National Forest land and flow down steep slopes to form narrow valleys containing almost all development. Overall impervious coverage in these watersheds is low relative to other study areas, although the local degree of impervious ranges up to comparable levels. In further contrast to the other regions, runoff in Vail is mostly generated by snowmelt, and relatively coarse soils are more infiltrative there. Local municipalities do not use formal structural BMPs at all and manage mainly with the non-structural strategy of riparian buffer maintenance.

The Puget Sound program quantified stream riparian characteristics during the period 1994-1997. Full geographic information system (GIS) coverage was not available in the many local jurisdictions where the streams are located, and the characterization was performed using aerial photographs and field reconnaissance (May 1996). To examine the relationship of riparian zones and stream ecology in the other three areas, their riparian zones were characterized using GIS data that had become available by the period 2000-2001. These analyses involved defining bands of specified widths on both sides of stream channels and quantifying various kinds of natural and developed land cover in these bands, as well the number of anthropogenic riparian corridor breaks per unit stream length. The main product of interest from each analysis was a data set representing buffer continuity and the linear extent of riparian buffers of various widths in several vegetation cover types.

An Index of Riparian Integrity (IRI) was developed in a manner similar to the B-IBI formulation (Fore, Karr, and Wisseman 1996) to express with one number the key attributes of riparian zones. Scores of 1 to 4, representing poor to excellent ratings or riparian buffering, were assigned to six attributes according to two measures of the lateral extent of the buffer, human encroachment into the buffer, corridor continuity, and

two measures of the riparian vegetative cover. The six scores were summed and divided by the total possible score to express the IRI as a percentage of maximum value. Table 4 later in the report provides more detail on the IRI.

The principal objective of the analysis was to compare patterns among the study locations of aquatic biological response to urbanization and the retention of watershed forest and wetland cover and stream riparian buffers. To permit comparison among study regions, invertebrate indices in each case were converted to percentage of the maximum possible score for the location. The coho salmon:cutthroat trout ratio (CS/CT) was an additional biological variable employed in Puget Sound data analysis. It was revealing in making these comparisons to plot biological measures against independent variables representing combinations of urbanization and the *de facto* non-structural BMPs. These variables were constructed to combine the hypothesized negative effects of urbanization (expressed as TIA) and loss of the non-structural elements (% watershed forest and wetland cover, Index of Riparian Integrity).

Results and Discussion—

Figures 1a to 1d present plots of biological measures versus one of the combination variables constructed to represent the watershed attributes, in this case multiplying the effects of impervious area, forest and wetland cover, and riparian integrity. Figures 1a to 1c for macroinvertebrates exhibit some quite consistent trends among regions that are discussed below. Vail data do not exhibit these trends, or other clear and consistent tendencies, and are not plotted. The differences in macroinvertebrate community responses in the Vail area compared to other locations, and the lack of clear relationships with urbanization, are likely due mainly to the small proportions of large watersheds that are developed there, as well as the unique physiography and terrestrial vegetation regime of the region. Analyses were performed using local measures of the independent variables, instead of watershed-scale measures, to see if aquatic biology associates more with nearby urbanization and natural land cover than overall watershed characteristics. These local measures represent land within 100 meters upstream and on each side of the stream measured from benthic macroinvertebrate sampling sites. Local TIA ranged as high as 26.0%, still substantially under maximum watershed TIA for other study locations. However, these analyses were not fruitful in discerning patterns helpful to understanding functioning of Vail area streams and managing them.

Figures 1a to 1d, along with the graphs for other combination independent variables not shown, exhibit several trends consistent among regions and ways of viewing the data:

1. The very highest biological indices in all cases are at extremely low values of the combination independent variables, meaning that in three different regions of the nation the best biological health is impossible unless human presence is very low and the natural vegetation and soil systems are well preserved near streams and throughout watersheds. These most productive, “last best” places can only be kept by very broadly safeguarding them through mechanisms like outright purchase, conservation easements, transfer of development rights, etc.

2. Biological responses to urbanization in combination with loss of natural cover do not indicate thresholds of watershed change that can be absorbed with little decline in health, the same as seen in the plots of biological measures versus TIA alone in earlier reports on this work (Horner and May 1999; Maxted 1999).

3. Regardless of location or variables considered, relatively high levels of biological integrity cannot occur without comparatively low urbanization and intact natural cover. However, these conditions do not guarantee fairly high integrity and should be regarded as necessary but not sufficient conditions for its occurrence.

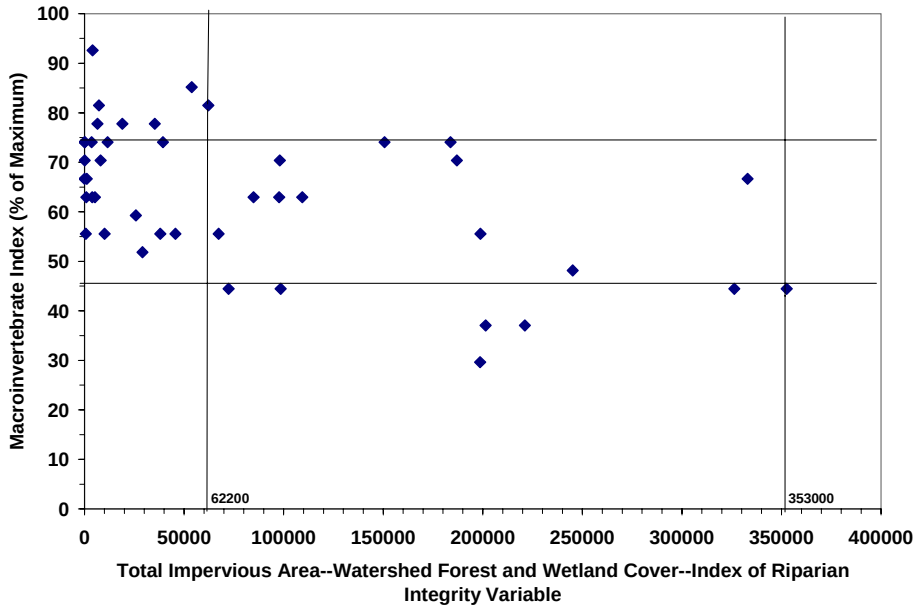
4. In contrast, comparatively high urbanization and natural cover loss make relatively poor biological health inevitable.

5. In all cases the rates of change in biology are more rapid to about the points representing crossover to relatively low integrity (the intersections of the lower horizontal and right-hand vertical line), and then further decline becomes somewhat less rapid. This pattern is probably a reflection of communities with organisms reduced in variety but more tolerant of additional stress.

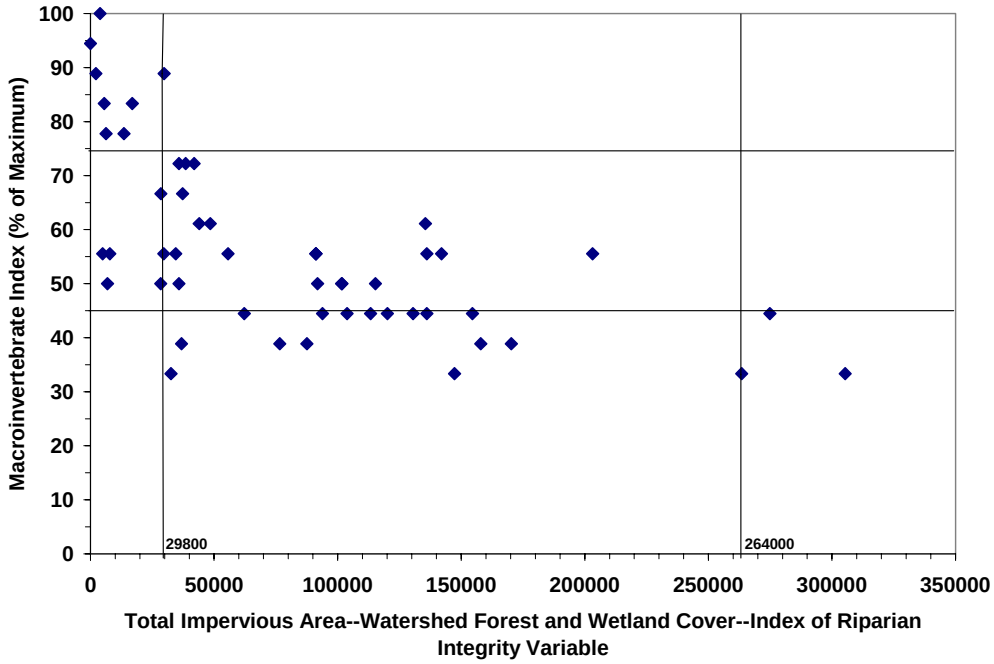
6. The points at which landscape condition takes away the opportunity for good biological health, or alternatively assures poor health, are similar among the study locations but deviate somewhat numerically. While these results might be put to general use in managing streams elsewhere, quantitative aspects should not be borrowed.

7. Comparing Puget Sound fish and macroinvertebrates, coho salmon exhibit more rapid rates of decline with landscape stress, lower points at which the quite healthy communities can exist, and also lower points of poor health.

In viewing these data, a reasonable question is whether or not protecting more forest and wetland, riparian buffer, or both can confidently be expected to mitigate increased urbanization. This question has considerable significance for the ultimate success of clustering development within low-impact designs to sustain aquatic ecosystems. In beginning to think about this issue, it must first be reiterated that if the goal is to maintain an ecological system functioning at or very close to the maximum levels seen, the answer is no. If the goal is to keep some lower but still good level of health, or to prevent degradation to a poor condition, though, the findings suggest that there is probably some latitude.

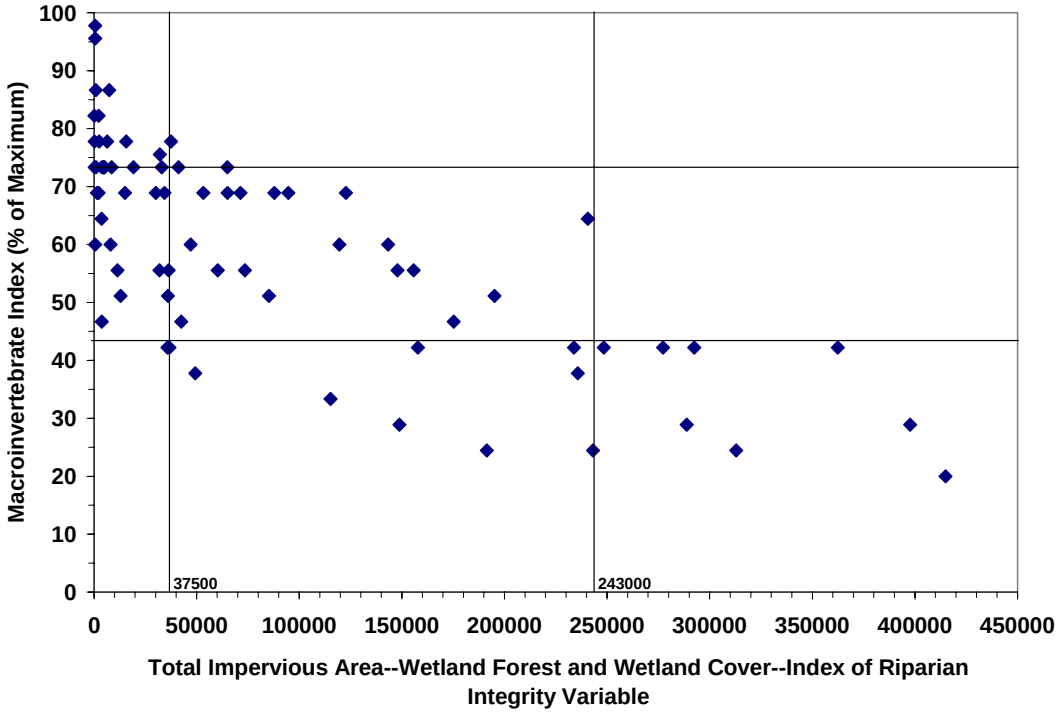


(a) Macroinvertebrate Index for Austin (*—sites atypically nutrient-enriched and omitted from analysis)

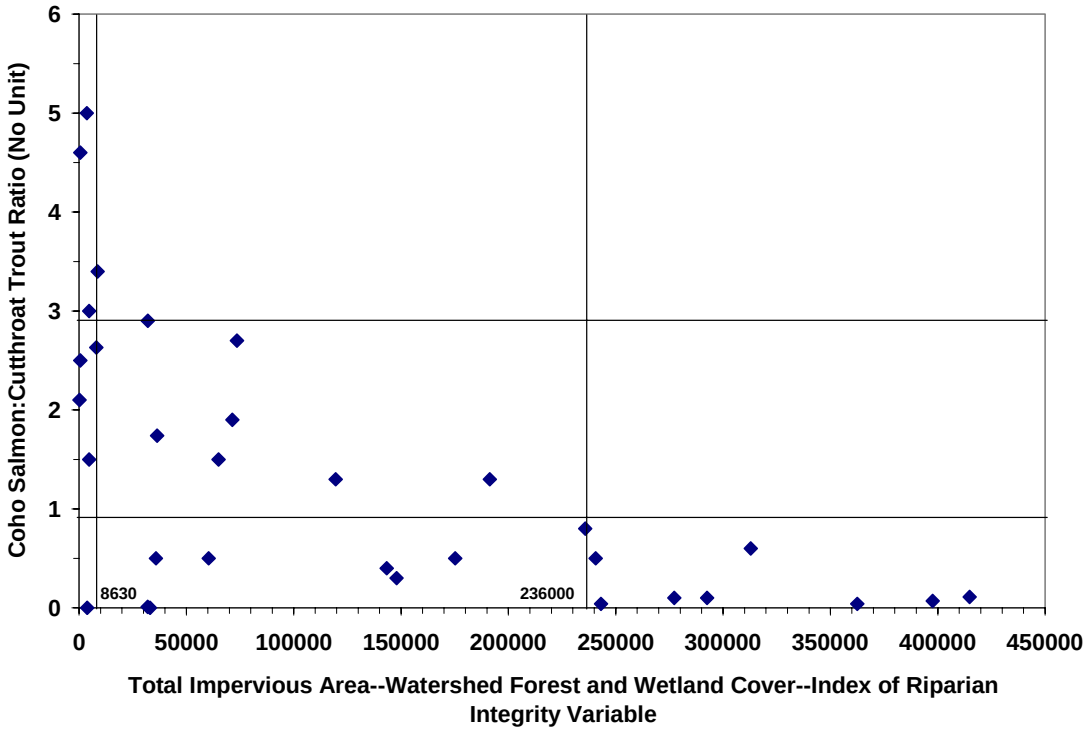


(b) Macroinvertebrate Index for Montgomery County

Figure 1. Biological Community Indices Versus (% Total Impervious Area, TIA)*(100 - Forest and Wetland Cover)*(100 - Index of Riparian Integrity, IRI) Variable [*Note: Upper and lower horizontal lines represent indices considered to d-fine relatively high and low levels of biological integrity, respectively. Left and right vertical lines indicate maximum TIA associated with high biological integrity and minimum TIA associated with low biological integrity, respectively. Numbers near the vertical lines are horizontal axis-intercepts.*]



(c) Macroinvertebrate Index for Puget Sound



(d) Coho Salmon:Cutthroat Trout Ratio for Puget Sound

Figure 1 continued

In this case the answer to the question can be investigated by using the horizontal axis-intercepts in Figure 1 as bases for examining combinations of the landscape variables in relation to biological goals. For example, the left-hand intercept in Figure 1(d) represents the simple algebraic equation, 8630 = (% TIA)*(100 - % watershed forest and wetlands)*(100 - % IRI). That equation can be solved for any of the three landscape variables, which can then be numerically computed by substituting selected values of the other two. If, for example, the biological goal is to provide necessary conditions for a relatively healthy coho salmon population (CS/CT ≥3.0), and the question is how much forest and wetland retention is necessary with TIA = 10 percent and IRI = 65 percent, an estimate is:

$$100 - \frac{8630}{(\%TIA) * (100 - \%IRI)} = 100 - \frac{8630}{10 * (100 - 65)} = 75\%$$

At least with the present level of understanding and confidence, analyses like this should be used in management only with caution and as advisory tools, and not as strict quantitative determinants. It must be kept in mind that, for high biological goals, the result only indicates the possibility, and not the certainty, of achieving the goal. Furthermore, the combination of numbers is not necessarily equivalent to an actual case from the database. Some combinations give nonsensical answers; e.g., a sum of impervious land, forest, and wetlands above 100 percent. Biological response depends on many circumstances not reflected in this simple analysis, such as where the developed area is relative to the stream and drainage pathways to it, what type of activity occurs there, and specific qualities of the natural landscape units. There are clearly limits to how much forest, wetlands, and riparian buffer can be preserved around development, particularly with the space constraints at moderate and higher urbanization levels. With all of these many factors unaccounted for, these data should be used only with care that conservatively protects resources.

If these cautions are recognized and taken, though, the findings from this multi-region study can be employed by watershed planners and managers as approximate guides. The authors' hope is that their use will reduce instances of decision making without specific goals and consideration of the most crucial elements that determine their achievement. Decisions made in this way should reduce simplistic, overly optimistic approaches that very often lead to resource deterioration. Meanwhile, research continued to an additional phase to develop models encompassing more components of complex watershed systems.

The best and safest use of the results is probably to analyze how to prevent deterioration to lower biological integrity, or to improve health somewhat, at medium to high urbanization. For one reason, the stakes are lower in this situation, as losses have already been sustained and the relatively tolerant organisms remaining are more robust than in more pristine areas in resisting change, should decisions result in unwanted outcomes. Also, the data show more certainty there than at lower urbanization, where favorable conditions are only necessary but not sufficient for predicting good health. Analyses were performed on some cases from this part of the urbanization spectrum in

the three regions, based on calculations like the one in the example above. The analyses demonstrated that, for the most part, staying above what has been defined as poor aquatic health requires holding TIA under 50 percent at usual levels of natural cover retention, or 60 percent with aggressive forest protection (about 5 percent lower in each case for Puget Sound salmon).

Detailed Puget Sound Structural BMP Assessment

Introduction and Methods—

Specific, direct evidence of the effectiveness of stormwater structural BMPs in protecting aquatic biota and receiving water beneficial uses is extremely sparse. As pointed out earlier, the few data do not give confidence in a clear biological payoff for the investments being made in these facilities, but are in no way adequate to warrant any solid conclusions in this regard. To add to this minimal information base, the Puget Sound component of the USEPA and WMI study conducted an intensive BMP assessment in the watersheds of four of its stream reaches, two in Big Bear Creek and one in its tributary Cottage Lake Creek (King County, WA), plus one in Little Bear Creek (Snohomish County, WA). Having received extensive management attention because of its rich salmonid fauna, the Big Bear Creek system has relatively large numbers of structural BMPs for its development level. The Little Bear Creek reach has relatively few structural devices for the urbanization level. Sites were divided in this way because of the observation in earlier work that BMP service level (density of coverage) varied widely among the urban catchments in the study and, as seems logical, is a factor in effectiveness. These five catchments contain a total of 165 individual BMPs, about 6.5 percent of the more than 2500 found in the entire regional survey.

All BMPs were located and visited in the field, where, if above ground, their dimensions were measured and various observations were recorded. For BMPs intended to control runoff water quality (wet ponds and biofiltration swales and strips), observations included vegetation cover, erosion, and sediment deposition. Maintenance condition was noted in both quantity and quality control facilities. King and Snohomish County stormwater management agency files had information on almost all of the BMPs, which supplemented the field data collection and observations.

The assessment went beyond service level to encompass quality of implementation as well. Implementation quality was rated according to a BMP Performance Index developed for this purpose. The indexing system encompasses structural BMPs designed to control the quantity of stormwater runoff (generally, peak flow rates) or its water quality, as well as those intended to serve both purposes. Appendix B contains the rationale for the index and the full procedure to compute it. The following paragraphs summarize the quantity and quality control elements.

Quantity control BMPs (mostly dry detention ponds and below-ground tanks and vaults, plus a few infiltration facilities) were rated in terms of their estimated replacement of natural soil and vegetation storage lost in development. Before development, the

watersheds were mostly covered by mature, second-growth forests almost entirely on till soils of glacial formation. For example, the Big Bear Creek site 4 catchment had >90 percent forest and wetland cover in 1985, when TIA was about 1 percent. Such conditions have been estimated to provide storage capacity for 15 to 30 cm of rainfall (Booth 1991; Booth, personal communication). Based on other local work on the till soils by Burges et al. (1998), 60 percent of this storage was estimated to be lost in the pervious portion of developed areas, and all would be lost in the impervious part. Storage replacement by infiltration devices was estimated as the volume that can be infiltrated in 24 hours as a function of the infiltration surface area provided and expected soil hydraulic conductivity. The volume detained in live storage for controlled release was taken as the replacement provided by ponds and under-ground facilities. It is recognized that, except for infiltration devices, the designs employed in these catchments are capable only of regulating peak rate discharge and not total volume ultimately released. Thus, they do not truly replace lost soil storage but only affect discharge patterns. An overall score of 100 percent for a catchment represents complete storage of all runoff from developed areas either via infiltration in 24 hours or in detention live storage.

For runoff treatment BMPs, implementation quality was gauged according to recognized design and maintenance standards for maximizing performance, which were expressed as condition scores. For wet ponds, the score was constructed according to wet pool volume relative to estimated design rainfall event runoff volume, ability to resist flow short-circuiting through flow path length and cellular configuration, emergent vegetation cover, and maintenance condition. For biofilters, the score depended on size in relation to the estimated amount needed to provide sufficient hydraulic residence time to achieve known performance capabilities, favorable slope, energy dissipation, vegetation cover, and maintenance condition. Scores were proportioned based on the consensus capabilities of the devices to remove two pollutants (total suspended solids and total phosphorus) and the amount of developed area served by each facility. Individual BMP scores were then added to compute an overall score for the catchment. A score of 100 percent represents interdicting all pollutants expected to be in design storm runoff from developed catchments, performance that could realistically be achieved structurally only by complete runoff infiltration.

Profile of Catchments and BMPs—

Table 1 summarizes the characteristics of the catchments and BMPs given detailed attention. Watersheds are as much as two-thirds developed but largely with medium-density single-family residences, producing TIA in or near the 5 to 10 percent range. The Big Bear and Cottage Lake Creek watersheds have the greatest coverage with structural BMPs among the 38 studied in the regional project. However, only about one-sixth to one-third of the developed area even has quantity control BMPs, the primary management objective in these salmonid streams subject to habitat destruction by more frequent elevated flows after urbanization. The average facility was built before the mid-1980s in the Cottage Lake Creek watershed, where many BMPs are below

ground. Those serving Big Bear Creek average 5 years younger and tend more to be surface ponds.

The quality control service levels are even lower, especially in the older Cottage Lake Creek developments (<5 percent of developed area). The much higher numbers in the Big Bear Creek catchments indicate the turn to quality control along with quantity control in the heavy development period there around 1990. The wet pond is the most prominent BMP type, somewhat exceeding biofilters in numbers. Most wet ponds perform double service as quantity control ponds with live storage too. Many installations are wet pond-biofiltration swale treatment trains, with ponds usually but not always draining into swales. Facilities expressly designed to be infiltration devices are relatively uncommon in these glacial till catchments.

The Little Bear Creek catchment has less coverage of developed areas by both quantity and quality control BMPs compared to the other watersheds. These cases thus provide a contrast in management under comparable urbanization.

Analysis—

Table 2 summarizes scoring of implementation quality for the two categories of BMPs. The analysis shows that < 4 percent of soil and vegetation storage lost to development was recovered by BMPs in the Cottage Lake and Big Bear Creek catchments, and approximately 1 percent in the Little Bear Creek cases. These very low percentages are in strong contrast to the proportions of developed areas having quantity control BMP storage, which are about an order of magnitude greater, although still far from complete. This dichotomy signifies inadequate standards for designing these BMPs, a point discussed further below.

Achieving the full potential of water quality treatment was similarly low. The Cottage Lake Creek catchment scored near the Big Bear ones despite a much lower service level because of substantially more infiltration there, a factor also reflected in its quantity control score.

This investigation started out to examine if the highest BMP service levels make a demonstrable difference in stream biological integrity. However, the mitigation potential provided by even these service levels proved to be so small that this question still cannot be conclusively answered. Biological measures are indeed lower in the relatively less served Little Bear Creek catchment, but factors other than structural BMPs could be responsible. Table 3 summarizes these potential factors for the five intensively studied catchments and two others with similar development but no structural BMPs at all. All of these streams are still producing salmon (generally, several species) and are thus resources to which strong management attention should be directed.

Table 1. Characteristics of Watersheds in Detailed Structural BMP Assessment

Characteristic[a]	Cott-2[b]	BiBe-1[b]	BiBe-4[b]	LiBe-2[b]
Catchment:				
Catchment area (km^2)	17.5	9.5	29.5	16.9
% developed	66.8	44.0	50.0	67.8
% impervious	11.1	6.6	8.3	9.9
Quantity Control BMPs:				
No. Qn BMPs	56	22	59	17
% Qn BMPs below ground	41.1	9.1	32.2	11.8
% developed area with Qn BMPs	30.9	24.2	15.9	11.5
Average age Qn BMPs (y)	13	8	8	9
Quality Control BMPs:				
No. Ql BMPs	11	22	49	5
No. infiltration devices	4	3	3	0
No. wet ponds	5	11	25	5
No. wet ponds that are also Qn BMPs	4	9	24	4
No. biofilters (swales, filter strips)	2	8	21	0
% developed area with Ql BMPs	4.6	15.4	13.5	3.4
Average age Ql BMPs (y)	11	8	7	9
Quantity and Quality Control BMPs:				
Total no. BMPs	63	35	84	18
Stream Biology:				
Benthic Index of Biotic Integrity	33	29	33	25
Coho Salmon:Cutthroat Trout Ratio	2.9	5.0	3.4	1.7

[a] Qn—quantity control; Ql—quality control; average ages are at time of stream ecology work; infiltration devices considered to be both quantity and quality controls; individual BMPs total 165, but table numbers do not sum to that total because some have combined functions and upstream BMPs also serve downstream stations.
[b] Cott-2—Cottage Lake Creek site 2; BiBe-1,4—Big Bear Creek sites 1 (upstream) and 4 (downstream); LiBe-2—Little Bear Creek site 2.

Table 2. Scoring of Quantity and Quality Control BMP Implementation

Score	Cott-2[a]	BiBe-1[a]	BiBe-4[a]	LiBe-2[a]
Quantity control score (%)[b]	2.0-3.9	1.5-3.0	1.2-2.4	0.8-1.6
Quality control score (%)	3.5	3.6	2.5	0.7

[a] See Table 1 note b.
[b] First number in range is score with assumption of maximum natural soil and vegetation storage (30 cm); second is with assumption of minimum natural soil and vegetation storage (15 cm).

Table 3. Watershed and BMP Conditions and Stream Biological Integrity in Eight Cases with Total Impervious Area in the Approximate Range of 5 to 10 Percent

Condition[a]	Cott-2[b]	BiBe-1[b]	BiBe-4[b]	LiBe-2[b]	GrCo-2[b]	LiSo-1[b]
TIA (%)	11.1	6.6	8.3	9.9	7.8	6.3
B-IBI	33	29	33	25	33	23
CS/CT	2.9	3.4	5.0	1.7		
% forest & wetlands	33.2	56.0	50.0	32.2	76.5	69.3
IRI	55.5	87.5	79.2	45.8	79.2	33.3
Qn score	2.0-3.9	1.5-3.0	1.2-2.4	0.8-1.6	0	0
Ql score	4.1	5.4	4.2	0.7	0	0

[a] TIA—total impervious area; B-IBI—benthic index of biotic integrity; CS/CT—coho salmon:cutthroat trout ratio; IRI—index of riparian integrity; Qn—quantity control; Ql—quality control.
[b] See Table 1 note b; also, GrCo-2—Green Cove Creek site 2; LiSo-1—Little Soos Creek site 1.

Table 3 does not present an entirely consistent picture. The Green Cove Creek reach equals the highest B-IBI among these sites without structural BMPs but has high levels of forest, wetlands, and riparian buffer preservation. The LiBe-2 and LiSo-1 sites exhibit the lowest B-IBI values and also substantially lower riparian indices than the other locations. Still, Cott-2 also equals the highest B-IBI with the highest and oldest development, nearly the least forest and wetlands, and only moderate IRI. It cannot be dismissed that this system is holding its level of health with the contribution of structural BMPs, even with their overall low service level and quality of implementation. Big Bear Creek has been the beneficiary of a King County program of fee-simple and conservation easement purchases that has encompassed 10.4 and 3.6 percent of the BiBe-1 and 4 catchments, respectively. These efforts are undoubtedly contributing to the thorough riparian buffering and moderate forest and wetlands retention seen there. Still, in biological measures these sites do not rise above the nearby Cottage Lake Creek catchment, which has very little (0.2 percent of the catchment) of these protected lands.

What is probably the safest observation is that many sources of natural variation in these ecosystems make clear-cut definition of cause and effect elusive. However, the general conclusion of the primacy of riparian buffering drawn in the preceding section appears to be upheld by these observations, and structural BMPs cannot be dismissed as contributing. Verification of that premise and delineation of how much protection they can actually afford requires their thorough and high quality implementation and then follow-up ecological study.

Discussion—

The analysis determined that, even in the watersheds around Puget Sound best served by structural BMPs, a distinct minority of the development has any coverage at all. The existing BMPs mitigate very small percentages of the hydrologic and water quality changes accompanying urbanization. To understand how this situation came, about it is worth reviewing some history of stormwater management in King County, which has jurisdiction over the relatively well served watersheds.

Agency records show the first detention ponds appearing in 1975. The first King County stormwater management regulation aimed at protection of aquatic ecosystems came in 1979. From the beginning of regulation, exemptions from compliance existed for relatively small developments (e.g., no requirement unless the development would create at least 5000 ft^2 of impervious surface). Many development projects are single dwellings or small short plats fitting in the exempted category. Exemptions largely explain why much of the developed area has no structural BMPs.

The 1979 regulation specified peak rate control ponds on the basis of a hydrologic estimation procedure based on the Rational Method. This rather crude procedure produced very inadequate pond sizes relative to vegetation and soil storage losses. These inadequacies resulted from the tendency of the method to underestimate pre-development discharges, which gave an artificially low target for post-development controls. Overall, detention ponds designed in this way recovered under 10 percent of the estimated lost vegetation and soil water storage (Booth, personal communication). These ponds thus gave very little water quantity control and, without any provisions for runoff treatment, no water quality mitigation.

A new King County regulation based on an improved method for hydrologic analysis (Santa Barbara Unit Hydrograph) took effect in 1990. This regulation also introduced water quality control requirements for the first time. Peak rate control ponds designed under it can replace perhaps two or three times as much lost storage as the preceding method (Booth, personal communication), an amount that still represents a small minority of the natural storage capacity. However, applicable law vests development applications filed before adoption of a new regulation at the standard prevailing at the time of application. In the rapid urbanization climate in the area *circa* 1990, many applications came under the old standard well into the 1990s. As a result, the large majority of the facilities in place when the stream ecology surveys were performed (1994-1997) were based on the very inadequate 1979 design criteria. Continuing deficiencies in design standards largely explain why, even where they are present, the facilities mitigate so little of the impact. These dual regulatory inadequacies of widespread exemption and insufficient implementation standards make inevitable the small beneficial effect of structural management, even where valued resources get a relatively high level of attention.

Relationship of Structural and Non-structural BMPs—

Stormwater and urban water resources management first developed around the concept of structural BMPs but recently broadened to encompass principles often given names like conservation design and low-impact development. Most fundamentally, these principles guide where to place development and how to build it to minimize negative consequences for aquatic ecosystems. There are many specific tools to implement them, but they fit generally into the broad categories of separating development from water bodies (i.e., retaining riparian buffers); limiting impervious area in favor of natural vegetation and soil, especially forest cover; and strategic and opportunistic use of structural BMPs. The Puget Sound database offers some opportunity to examine how these structural and non-structural strategies might fit together and what they can accomplish in different urbanization scenarios.

Figure 2 encompasses the various general elements of conservation design and how they relate to stream biology in terms of macroinvertebrates. Structural BMPs are expressed as the density of BMP coverage per unit area of impervious surface (sites with TIA <5 percent do not have structural BMPs and are excluded). Non-structural practices are represented as the product of watershed forest and wetland cover (percent) times index of riparian integrity (percent of maximum) and graphed for the highest, intermediate, and lowest one-third of the resulting numerical values.

The first observation that should be made about Figure 2 is that the five highest macroinvertebrate indices are not represented, because they are from sites with <5 percent TIA. It is apparent that neither structural nor non-structural measures, at least at the levels represented in this database, can provide for the highest benthic macroinvertebrate integrity if any but the most minimal development occurs.

It can further be observed in Figure 2 that points at the left (relatively few BMPs) disperse widely over the macroinvertebrate index range. Some sites with little forest, wetland, and riparian retention rise into the intermediate biological integrity zone (45 to 75 percent of maximum index value), while a few locations with higher non-structural measures fall close to or into the region of relatively low ecological health. This observation is an expression of what is also apparent in Figures 1a to 1d, namely that a certain ecological status is not assured by any condition, or even combination of conditions, but is only more likely with those conditions.

The Figure 2 points converge with increasing structural BMP density, overall and in each non-structural category. Sites with the lowest macroinvertebrate indices (and also highest urbanization and lowest non-structural measures) appear to benefit from structural BMP application. Those with higher biological and natural cover measures and lower urbanization do not, with the result that points tend toward the intermediate biological level. If ecological losses are to be stemmed at high urbanization, structural BMPs appear to have a substantial role. In this situation development has taken forests and wetlands and intruded into riparian zones, reducing the ability to apply non-structural options.

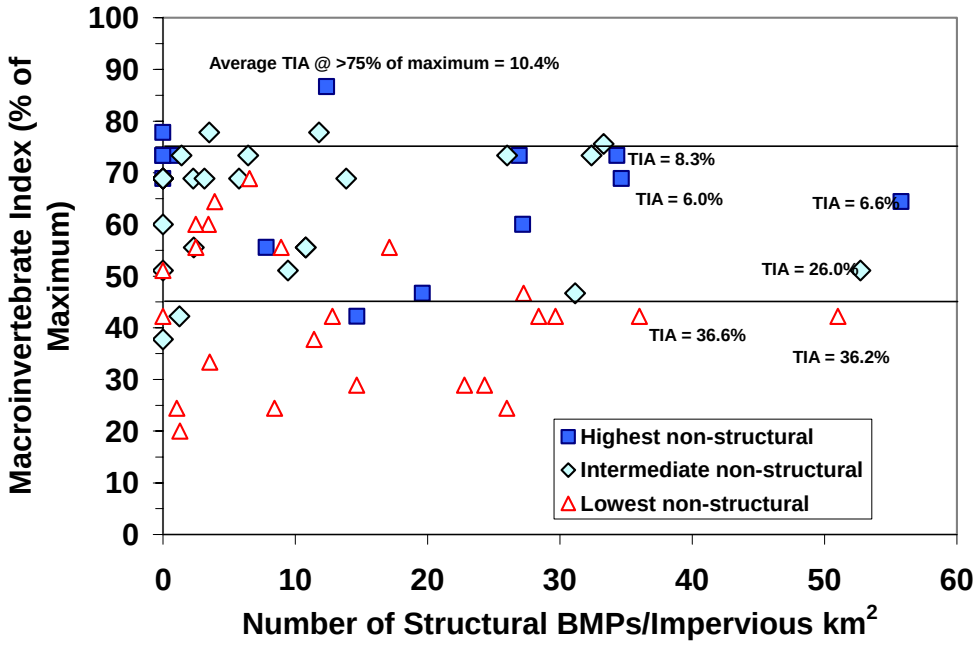


Figure 2. Macroinvertebrate Community Index Versus Structural BMP Density with the Highest, Intermediate, and Lowest One-Third of Natural Watershed and Riparian Cover [*Note: Upper and lower horizontal lines represent indices considered to define relatively high and low levels of biological integrity, respectively.*]

Any conclusions from this analysis must be tempered according to the scope of the underlying data. Probably the leading factor giving caution is that no instances exist of structural BMPs being exceptionally widely applied and designed to mitigate a large share of the known impacts of urbanization. Therefore, the fullest potential of these practices has not been examined, and it is possible that extremely thorough applications would demonstrate additional benefits not suggested in these data.

ADDITIONAL RESEARCH TASKS (PHASE 4)

General Research Needs and Objectives

Upon completion of Phase 3 of the US EPA and WMI project, WMI and EPA staff met to discuss the project's results and the most fruitful avenues to conclude work under the cooperative agreement in relation to present US EPA needs. The areas identified, which form the research core of the Phase 4 work, were:

- Obtain consensus among practitioners from the study regions on how watershed assessments for studies of this type should be performed, determine how

geographic information systems (GIS) should be developed to provide a foundation for these analyses, and prepare a protocol to guide GIS-based watershed assessments.

- Perform additional analyses on the existing data set, especially to improve definitions of pervious and impervious land cover and the relationships linking landscape quality and human activities with receiving stream biological integrity.

Development of GIS-Based Watershed Assessment Tools (Phase 4 Task 1)

In the past five years GIS has become a commonly used tool in state and local government agencies, even relatively small ones. The spread of this technology has greatly eased the tasks of archiving, transferring, and analyzing watershed data. This project has made extensive use of GIS databases from Austin, Montgomery County, and Vail and brought Puget Sound data into that form under Phase 4. However, disparities in the form and coverage of the various data sources impede efficiency and effectiveness in using the data. In the earlier phases the collaborators in this project gained substantial experience in putting the GIS databases to work toward the objectives and developed ideas about what they should contain and how they should be packaged to assist this type of work. The main objective of this task was to put the ideas together to reach a consensus and then to translate that concept to guidance that can be distributed to others who undertake similar work. Common methods will allow accumulation of widespread data that can be applied in other locations to test the conclusions reached in this work for application elsewhere. The next section of the report extensively details the development and application of the GIS-based watershed assessment tools for the three regions.

Additional Analyses of Existing Data (Phase 4 Task 2)

Analyses during Phases 1-3 yielded substantial insights on the functioning of stream ecosystems in relation to watershed conditions and information that can be applied to improve watershed management. These conclusions were built largely on general independent variables that aggregate watershed attributes: (1) total impervious area, representing development; and (2) proportion of the watershed in forest and wetland cover, representing pervious land best preserved in a natural state. The earlier analysis was able to get more specific with riparian zone definition, having developed an index of riparian integrity that assimilates six characteristics representing the lateral and longitudinal extent of vegetated riparian land, the quality of the vegetation cover, and its continuity (freedom from crossings and intrusions by human works and activity).

The first objective of this task was to follow up the progress with riparian zone definition by using GIS information from the first Phase 4 task to bring more specificity to characterization of watershed pervious and impervious cover. Impervious cover was more closely defined in land use terms such as residential occupancy of various densities, commercial, light and heavy industrial, major transportation corridors, etc. The intent with this classification is to represent not only the hard surface but also the

character of these lands, and the activities that occur there, affecting receiving waters. The different types of pervious cover were distinguished in terms of type of vegetation.

The definitions of pervious and impervious cover were applied to both the overall watersheds and riparian zones of different extension from the stream centerline (e.g., 10, 50 and 100 meters on each side). For Austin and Puget Sound the available GIS data also allowed characterization of “local” zones (e.g., a full circular arc 300 meters in diameter extending up-gradient from the stream sampling point).

The work then returned to the analyses performed on the Austin, Montgomery County, and Puget Sound databases with these more complete characterizations to see if they reduce dispersion seen in the data, especially at relatively low urbanization, and give a more incisive functional portrait and interpretations that can further improve management guidance. These analyses employed various graphical, statistical, multivariate, and indexing techniques. Succeeding sections specify the methods by which these techniques were applied and then present and discuss the results.

WATERSHED ASSESSMENT PROTOCOL

INTRODUCTION

The goal of the first task in Phase 4 was to develop a watershed assessment protocol to provide guidance for performing a quantitative assessment of the ecological health of a watershed based on landscape-level characteristics. The resulting protocol also provides a road map for developing a relative risk model for a watershed using inputs from stakeholders. GIS analytical techniques are used to relate multiple parameters that potentially impact the ecological health of the watershed. The assessment uses the principles of landscape ecology, which takes into account the spatial arrangement of the components or elements that make up the environment. Landscape ecology recognizes the relationships among ecological patterns, including both humans and their activities as integral parts of the environment. Because watershed ecosystem condition depends on a large number of landscape factors, this protocol describes the methods for measuring and ranking these factors using accepted watershed assessment methodologies and standard GIS analysis procedures.

General Watershed Assessment

Watershed assessment is an on-going process of compiling and analyzing technical information on watershed conditions and the effects of human activities on those conditions. It is typically one of the first steps in a long-term ecosystem management program. Ultimately, watershed management, including conservation and restoration activities, will be a key component in any integrated regional ecosystem management effort. Because data availability and technical resources vary across geographic areas, the way watershed assessment tasks are approached and the time frame for accomplishing them varies. A single assessment method or tool will likely not provide all of the information needed to manage effectively a diverse watershed. However, a standardized assessment framework is needed to bring together different assessment elements, improve consistency, and synthesize data for more effective watershed management.

A science-based watershed assessment analyzes the current state of the watershed, captures its unique physical, chemical, and biological characteristics, and compares these conditions to those in historic, natural, or “reference” watersheds. It should explicitly identify uncertainty of information and be supported by written records that provide a basis for decision-making. Assessments help determine the structure of the watershed aquatic ecosystem, how well the watershed is functioning, and how it responds to natural and human disturbances (see Figure 3). This process should help in better understanding how a watershed “works” and how a watershed has changed as a result of human activities.

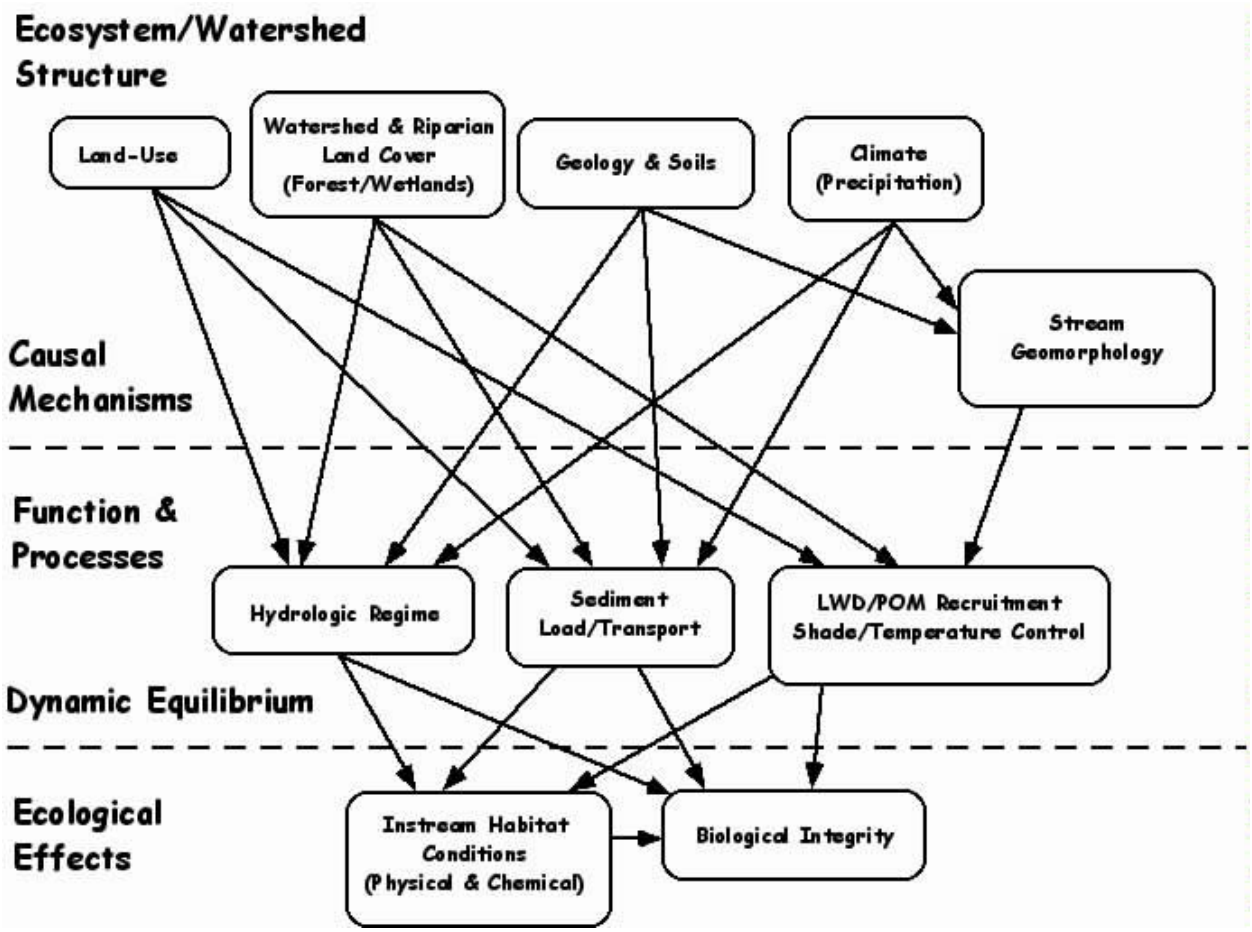


Figure 3. Conceptual Diagram of Watershed Structure and Function (Anderson et al. 1976)

The key components generally involved with watershed assessment include the following (U.S. Environmental Protection Agency 2000):

1. Describe the <u>general physical, biological, and chemical attributes of the watershed</u> (i.e. geology, climate, topography, hydrology, and soil structure). This can be done at a qualitative or quantitative level.

2. Identify and describe <u>factors within the watershed</u>, both natural and human-caused, that affect the physical, biological, and chemical attributes of the watershed. This includes analysis of:

    - Hydrologic conditions—low flows, peak flows, water use, land cover, land-use, and impervious surfaces;
    - Soil erosion and sediment load and sources—roads, construction sites, landslides;
    - Natural vegetation patterns and characteristics—riparian corridors, upland

forests, and other native vegetation;

- Wetland, floodplain, and nearshore/estuarine conditions within or adjacent to the watershed;
- Biological communities—aquatic and terrestrial biota; and
- Water quality conditions—chemical characteristics and pollutants of concern.

3. Identify and describe human-caused factors existing outside of the watershed that affect the physical, biological, and chemical attributes of the watershed; (e.g., an upstream dam that regulates flow or modifies water temperature, a source of chemical pollution or turbidity, or degraded aquatic habitat).

4. Classify and subdivide the watershed into sub-watersheds to facilitate comparative assessment of existing conditions and prioritization of management prescriptions.

5. Identify data gaps that need to be filled to reduce uncertainty in the development of recommendations to address problems.

As a general principle, protection of intact, functional watersheds should be a high priority (National Research Council 1992, 1996; Beechie and Bolton 1999; Roni et al. 2002). Maintaining high quality habitat is typically much easier than restoring degraded habitat (Roni et al. 2002). These protected watersheds can also serve as "reference areas" or "templates" for restoration of already degraded habitat. Such areas can provide fundamental information on natural processes and ecosystem structure, enable evaluation of temporal and spatial changes, and provide a basis for estimating attainable future conditions for restoration (Hughes et al. 1986). Protection of aquatic resources involves preventing anthropogenic alterations to the structure and function of the ecosystem (National Research Council 1992). Ideally, this effort would involve protection of biological, chemical, and physical processes that maintain salmonid habitat. The mechanisms for protection include regulation, acquisition, conservation easements, and stewardship.

In contrast, restoration involves improving aquatic functions and related physical, chemical, and biological characteristics of riparian and aquatic systems. Restoration may involve reconstruction of previous physical conditions, adjusting physical and chemical processes, removal of invasive or exotic species, and/or biological re-introduction of extirpated species (National Research Council 1992). Aquatic restoration activities also may include improvements in water quality, water quantity, fish passage, and riparian vegetation. Different types of restoration activities vary in the amount of human intervention involved and represent a continuum of "passive" to "active" restoration (National Research Council 1992). The term restoration is typically used broadly to encompass activities that are sometimes classified as "rehabilitation", "enhancement", or "mitigation" (National Research Council 1992). Figure 4 illustrates these concepts. As used here, watershed restoration is defined as the process of

restoring systems and processes to the point they can provide the natural materials and ecological functions that create functional habitat and support native biota.

Rehabilitation (Figure 4) has been defined as re-establishment of naturally self-sustaining or properly functioning aquatic-riparian ecosystems to the extent possible while acknowledging that irreversible human changes might permit only partial restoration of ecological functions (National Research Council 1996). In most areas, the restoration of pre-European settlement watershed conditions and processes is not usually feasible because of societal constraints and/or irreversible landscape changes. However, at individual sites or in specific sub-watersheds, full restoration may be possible.

Enhancement (Figure 4) focuses on improving selected habitat characteristics, and may involve technological approaches or artificial structures that mimic habitat elements, as well as natural methods (National Research Council 1992). Mitigation describes a range of activities with the intent to avoid, reduce, or compensate for the effects of environmental damage (National Research Council 1992). Based on this definition, one can see that mitigation is not designed to be 100 percent effective in protecting natural ecosystem structure and function. Yet, “mitigation” is the actual basis of most of urban watershed and stormwater management strategies.

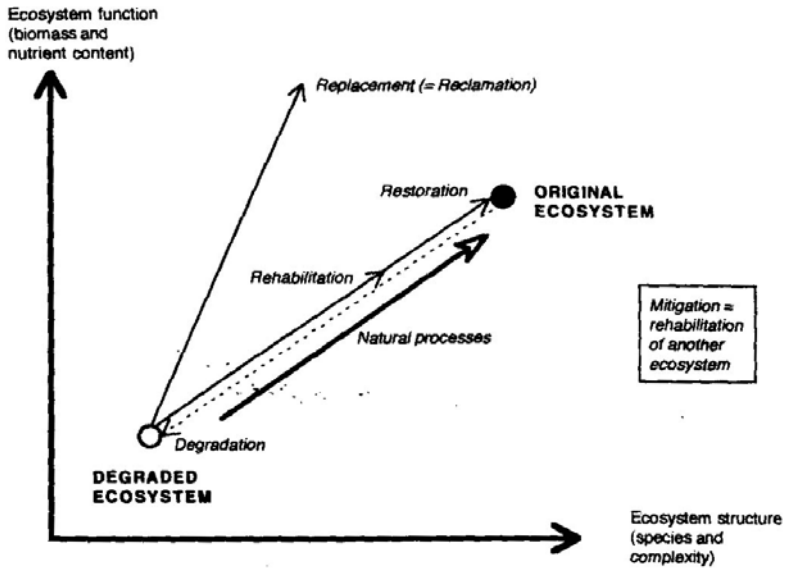


Figure 4. Resource management Concept Diagram (Bradshaw, 1996)

The type of protection and restoration activities can be prioritized according to their likelihood for longevity of response and probability of success. However, the effectiveness of various restoration techniques is not always well understood. Inherent variability in ecological data can make it difficult to detect real change. Long-term monitoring data are often needed to evaluate restoration success. Lack of effective

monitoring, including an emphasis on physical rather than biological factors, has limited understanding of aquatic ecosystem response to restoration. The location of protection and restoration activities can also be prioritized according to their probability of being effective. Prioritization can be conducted at multiple spatial scales such as basin, watershed, and sub-watershed scales. Many criteria may be weighed when prioritizing restoration and preservation of key watersheds (National Research Council 1992).

During prioritization of restoration activities within a watershed, landscape position is an important factor to consider. Some argue that restoration should proceed from headwater reaches downstream, largely because human disturbances proceeded from the river mouth into headwater areas. The successful preservation and restoration of downstream areas often depends on restoring upstream processes and upland land use. Minimizing adverse impacts on small headwater streams and restoring their ecological connections are of critical importance to the improvements downstream (National Research Council 1992). Often, the effects of disturbances such as floods are magnified as they progress downstream. In addition, stream restoration without modification of up-slope land-uses could be ineffective (Doppelt et al., 1993).

Stream Riparian Zone Assessment

In investigations of relationships between stream ecosystem and the surrounding landscape, an analysis of riparian conditions should be performed in conjunction with the overall watershed assessment. In the Pacific Northwest, it has also been observed that conserving an extensive, wide, and mature native forest buffer between developed areas and aquatic resources tends to maintain a substantially higher level of ecological function than is found where little or no riparian corridor is protected (May et al. 1997a, b). In addition, it has also been demonstrated that riparian corridor fragmentation by roads generally has a detrimental impact on stream ecosystems, water quality, and hydrologic function. Data from the Puget Sound region indicates that a nearly continuous forest corridor has a positive affect on in-stream habitat conditions, as well as water quality (May et al. 1997a, b). The results of both the watershed and riparian landscape-level analysis, along with any modeling, can provide the basis for management recommendations and restoration initiatives.

As was the case with the watershed landscape assessment, when analyzing the riparian corridor, it is important to take a holistic approach to the problem. A thorough understanding of ecosystem structure and function is necessary. In addition, natural or “reference” conditions must be established. This information allows comparison of current conditions with “target” conditions, allowing for both the evaluation and prescription phase of rehabilitation or restoration. The reference conditions selected need not be pre-European settlement conditions, but could be some alternative structural state that supports ecological function at a desired level. Using the Pacific Northwest (PNW) as a template, this section describes the concept of *riparian integrity* and how it can be measured.

In the PNW, a wide, nearly continuous corridor of mature forests, off-channel wetlands, and complex floodplain areas characterizes natural stream-riparian ecosystems of the region (Naiman and Bilby 1998). Native riparian forests are typically a mosaic of complex, multi-layered forests dominated by mature conifers, mixed with patches of alder where disturbance has occurred in the recent past (Gregory et al., 1991). The riparian forest also includes a complex, dense, and diverse understory and ground-cover vegetation. In addition, the extensive upper soil layer of forest "duff" provides vital water retention and filtering capacity to the ecosystem.

A typical PNW natural riparian corridor also includes a floodplain area, channel migration zone (CMZ), and numerous off-channel wetlands. Natural floodplains, the CMZ, and riparian wetlands are critical components of a properly functioning aquatic ecosystem (Naiman and Bilby,1998). Organic debris and vegetation from riparian forests also provide a majority of the organic carbon and nutrients that support the aquatic ecosystem food web in these small lowland streams. In short, the riparian community (vegetation and wildlife) directly influences the physical, chemical, and biological conditions of the aquatic ecosystem.

Reciprocally, the aquatic ecosystem affects the structure and function of the riparian community. Anadromous salmonids provide a rich, seasonal food resource that directly affects the ecological integrity of both aquatic and terrestrial food webs. The potential contribution of nutrients from decomposing salmon carcasses to the forest was historically quite significant (Cederholm et al. 1989; Knutson and Naef 1997; Willson et al. 1998; Cederholm et al. 1999). The presence of this seasonally abundant nutrient resource has also had a hand in shaping the evolution of the stream-riparian ecosystem (Willson et al. 1998). The aquatic ecosystems of the region are very closely linked ecologically with the surrounding terrestrial ecosystems. Therefore, any human impacts that destroy or degrade habitat in the stream-riparian corridor, or reduce the abundance and diversity of wildlife that are associated with anadromous salmonids, have the potential to weaken the ecological linkages in this complex ecosystem.

In addition to the characteristics of the riparian forest described above, the most commonly recognized functions of the riparian corridor include the following (Gregory et al. 1991; Naiman and Bilby 1998; Federal Interagency Stream Restoration Working Group (FISRWG) 1998):

- Providing canopy-cover shade necessary to maintain cool stream temperatures required by salmonids and other aquatic biota; regulation of sunlight and microclimate for the stream-riparian ecosystem;

- Providing organic debris, leaf litter, and other allochthonous inputs that are a critical component of many stream food webs, especially in headwater reaches;

- Stabilizing stream banks, minimizing stream bank erosion, and reducing the occurrence of landslides, but still providing stream gravel recruitment;

- Interacting with the stream channel in the floodplain and CMZ; retention of flood waters; reduction of fine sediment input into the stream system through floodplain sediment retention and vegetative filtering;

- Facilitating the exchange of groundwater and surface water in the riparian floodplain and stream hyporheic zone;

- Filtering and vegetative uptake of nutrients and pollutants from groundwater and stormwater runoff;

- Providing recruitment of large woody debris (LWD) into the stream channel; LWD is the primary in-stream structural feature that functions as a hydraulic roughness element to moderate stream flows; LWD also as serves a pool-forming function, providing critical salmonid rearing, refuge from elevated flows, and enhanced in-stream habitat diversity; and

- Providing critical wildlife habitat, including migration corridors, feeding and watering habitat, and refuge areas during upland disturbance events; providing primary habitat for aquatic habitat modifiers such as beaver and many other terrestrial predators or scavengers associated with salmonid populations.

The effects of watershed urbanization on stream riparian ecosystems are well documented. They include extensive changes in basin hydrologic regime, channel morphology, and physicochemical water quality. The cumulative effects of these alterations produce an in-stream habitat considerably different from that in which salmonids and associated fauna evolved. In addition, development pressure has a negative impact on native riparian forests and wetlands, which are intimately involved in stream ecosystem functioning. Much evidence of these effects exists from studies of urban streams in the Puget Sound region (Richey 1982; Scott et al. 1986; Booth and Reinelt 1993; Horner et al. 1997; May et al. 1997a, b).

During the last of these regional studies of urbanizing watersheds, it became apparent that so called riparian “buffers”, if designed and maintained to emulate natural riparian conditions, could have a significant mitigating influence on the ecological degradation of streams and wetlands in urbanizing watersheds of the Puget Sound Lowland ecoregion (May et al. 1997a, b). This influence was reflected in higher than expected levels of biotic integrity in those stream reaches with wide, continuous, and naturally vegetated buffers (including intact headwater and off-channel wetlands).

Based on the results of the initial Puget Sound stream studies and the resulting understanding of the structure and function of natural stream riparian ecosystems, the term, *riparian integrity* was adopted to describe the conditions found in these ecosystems. These “properly functioning conditions” should serve as a template for conservation and recovery of riparian areas. As used here, riparian integrity includes

both structural and functional elements characteristic of the natural stream riparian ecosystem.

Riparian buffer width or extent is often the sole criterion by which riparian corridor management areas are defined. Buffers can be “fixed” or “variable” in width. Fixed-width buffers are generally the products of political compromise between protecting a natural resource and minimizing the impact on development and private-property rights. Buffers of this type, unless conservatively designed and managed, often fail to support all the ecological functions of the riparian corridor. Variable-width buffers have the potential to be more ecologically based, but are difficult to administer by jurisdictions.

Impacts of human activities on riparian ecosystems are numerous and highly variable. The complex mosaic of land use in developing watersheds results in multiple stressors impacting the stream-riparian ecosystem. The characteristics of the riparian ecosystem will also influence the extent and intensity of the human-induced disturbance. Streams in watersheds dominated by rural development have different impacts than those in suburban or urban watersheds.

Variables such as stream size, location within the watershed, stream gradient, valley configuration, watershed topography, soil type, and others all combine to make some stream riparian ecosystems more or less sensitive to surrounding human impacts. For example, a stream with an extensive floodplain area or active CMZ will react quite differently than a stream within a deep, steep-walled ravine. It stands to reason, then, that appropriate buffer size depends on the spatial area necessary to maintain the desired riparian functions and on the land use activities influencing the stream-riparian ecosystem. For example, a wider buffer may be required in situations where high-intensity land use is found than in areas with less intense use of the land. Similarly, all else being equal, a sensitive tributary stream used by salmonids for spawning and rearing may require larger buffers than a main stem stream used only as a migration corridor. In general, urban riparian buffers have not been consistently protected or well managed (Schueler 1995; Wenger 1999; Moglen 2000).

Of equal importance to the width or extent of the riparian corridor is the quality of the riparian area in terms of vegetation type, species diversity, physical condition, and maturity. Ideally, the riparian corridor in a developing or developed watershed should mirror that found in the natural ecosystems of that region. As was discussed earlier, mature, coniferous-dominated forests with a mosaic of young forest and riparian wetlands characterize natural riparian corridors in the Puget Sound Lowland ecoregion. Due to the cumulative impacts of past and present land-use, this type of cover is often not found.

The riparian quality of the stream corridor, along with the width or extent, determines how well a particular riparian area performs all of the functions required of it. The current vegetative composition and maturity should be factored into any riparian management or buffer width design effort. Areas dominated by mature, naturally complex forest have a much higher conservation potential than disturbed areas, young

stands of native forest, or exotic vegetation. These mature and naturally diverse riparian areas also perform their required functions more efficiently and tend to be more resilient in recovering from disturbance (Naiman and Bilby 1998).

Past land-use activities in the Puget Sound region (timber harvest, road construction, and agricultural activities) have significantly impacted the riparian forests of the region (Horner and May, 1999). As a result, many riparian forests in urbanizing watersheds of this region are dominated by relatively young stands of alder and maple rather than the mixed mature forests that characterize natural riparian communities of the region (May et al. 1997a, b).

Riparian corridor connectivity is also an ecologically critical and often under-emphasized component of riparian integrity. Natural riparian corridors in the PNW are nearly continuous throughout the stream riparian ecosystem (Naiman and Bilby 1998). In addition to buffer width and quality, management of the riparian corridor should focus on minimizing fragmentation. Road crossings, utility rights of way, and other breaks in the riparian corridor effectively reduce the buffer width to zero and provide a conduit for runoff and pollutants to enter the stream (Schueler 1995; May et al. 1997a, b; Weller et al. 1998). Breaks in the riparian corridor should be kept to a minimum, and all breaks should be designed for minimal stormwater and other impacts (Horner and May 1999).

Floodplain connectivity is also critical to a properly functioning stream-riparian ecosystem (Gregory et al. 1991; Naiman and Bilby 1998; FISRWG 1998). Therefore, the active CMZ and floodplain should be included in the designated "riparian management zone" (RMZ). Both from an ecological and public safety perspective, development should be excluded from the RMZ. In general, encroachment of developing areas should be prevented from impacting the structure or function of the stream-riparian ecosystem. This goal can be achieved via public education and by clear delineation of the RMZ (Schueler 1995).

The above discussion uses the experience in the PNW to briefly illustrate the level of understanding necessary to fully evaluate the stream-riparian ecosystem. Based on the analysis of riparian parameters in comparison to ecological conditions, a set of "metrics" can be developed. The data generated from the riparian analysis can then be converted into an "*Index of Riparian Integrity (IRI)*." This index can then be used as a management tool for rating existing riparian conditions or evaluating the effectiveness of conservation activities.

The Index of Riparian Integrity should include metrics that measure each of the main components of natural riparian ecosystem integrity. These measures will vary depending on the ecoregion and the unique structural and functional elements of regional riparian integrity. However, there are some common characteristics that should be represented by regionally appropriate metrics. These characteristics include measures of the lateral extent of the riparian zone, attributes related to riparian quality or vegetation conditions, and some measure(s) of riparian corridor continuity or

fragmentation. It may also be advantageous to select multiple metrics for each of these main components of riparian integrity.

The metrics selected for the initial Puget Sound research were those that had the strongest correlation with in-stream habitat and biological monitoring (B-IBI) scores within each of the three main components of riparian integrity (riparian extent, longitudinal continuity, and vegetation quality). For other regions of the country, other riparian metrics may be selected for the Index of Riparian Integrity. In any case, it is important that the selection be based on the accepted ecological functions of the riparian zone and not on political restrictions or data limitations.

Six metrics comprise the Puget Sound IRI. After the metrics were identified based on correlation analysis relative to B-IBI, they were individually scored by placing them into one of the following categories with the associated metric score (see Table 4): (1) optimal—4, (2) good—3, (3) fair—2, and (4) poor—1. The individual scores for all indices were then added to produce a single IRI score. This number was divided by the maximum possible total to express the IRI as a percentage of the optimal score.

Table 4. Index of Riparian Integrity Metrics and Scoring Criteria

Index of Riparian Integrity Metric	Excellent (4)	Good (3)	Fair (2)	Poor (1)
Width (lateral extent > 30 m, %)	> 80%	70-80%	60-70%	< 60%
Width (lateral extent > 100 m, %)	> 50%	40-50%	30-40%	< 30%
Encroachment (% < 10 m wide)	< 10%	10-20%	20-30%	> 30%
Corridor continuity (crossings/km)	< 1	1-2	2-3	> 3
Natural cover (% forest or wetland)	> 90%	75-90%	50-75%	< 50%
Mature native vegetation or wetland (%)[a]	> 90%	75-90%	50-75%	< 50%

[a] "Mature" vegetation was considered to be the type, and in some cases average tree size (diameter at breast height, dbh), in the least disturbed reference sites, typical of natural riparian structure and functioning, even if not developed to the maximum extent that would be reached in more time. The specific definition is > 70 percent coniferous forest with dbh > 30 cm (12 inches) and native understory.

DEVELOPING GEOGRAPHIC INFORMATION SYSTEMS FOR WATERSHED ASSESSMENT

GIS Data Requirements

In its most general definition, a *watershed indicator* or "metric" is a number that is calculated by summarizing data that represent a specific condition within the watershed. Landscape indicators are measurements of ecological structure (such as the amount of forest cover) and ecological function or watershed processes (such as mass-wasting

potential or hydrologic regime). These indicators or metrics often correspond to a specific map-layer in the GIS.

In general, the land cover and land-use pattern within a watershed is a good overall indicator of the ecological conditions of the watershed. Forests often filter pollutants, preventing them from reaching the water, whereas urban land-use often contributes pollutants to streams and marine water bodies. Forests also dissipate energy associated with major rain events; this function reduces nutrient loading and reduces severity of flooding. A simple summary indicator might be the percentage of forest cover within a watershed. To refine this indicator, a scientifically valid assumption might be used to help account for "natural" conditions, for example what type of forest was the natural cover of the watershed. The landscape indicators that are selected are typically most representative of the ecological health of the watershed. The indicators can also be categorized into the following general classes of landscape ecology:

- Land-use or cumulative human impact indicators that reflect the potential impact of human activities on the ecological health of the watershed (e.g., imperviousness, road density, land-use);

- Hydrologic landscape indicators that reflect the potential impact of the hydrologic budget and factors that control the impact the watershed hydrologic budget has on the ecological health of the watershed (e.g., soils, slopes, forest cover); and

- Land cover and landscape change indicators that reflect the potential impact of land cover classes and land cover change on the ecological health of the watershed (e.g., percentages of wetlands lost, change in forest cover, increase in imperviousness).

The following GIS-based data sets are recommended to carry out a complete watershed assessment:

1. Hydrologic Layer—This layer is typically available from federal, state or local natural resource agencies. It typically contains all surface waters including rivers, streams, lakes, and wetlands, as well as any associated marine waters. The delineation of streams, especially small, headwater systems should be done at a level of detail that includes seasonal/ephemeral channels as well as the perennial streams typically delineated on USGS topographic maps. Delineation may require the use of a digital elevation model (DEM) or some other GIS algorithm based on topographic features of the watershed under study to "route" the stream channels accurately. Due to the importance of the stream network to the overall watershed assessment process, a high resolution, accurate hydro-layer is essential. In addition, field verification of stream channel locations, extent, and seasonal flow patterns will likely be required. Having a hydrologist as a member of the watershed assessment team is obviously a must.

Watershed delineation is also a required component of the overall process. The GIS analyst should delineate watershed boundaries to the level of detail required by the study. Typically, drainage basins must be delineated at the level of a small stream or "sub-watershed" level. To be useful at the scale necessary for studying or managing landscape-level changes caused by development, the level of delineation must be at least to the 6th level of hydrologic unit code (HUC) as designated by the U.S. Geologic Survey (USGS). It is recommended that delineation be to the HUC-8 level if possible. One of the most important considerations for watershed delineation is that the "pour-point" upon to which the boundaries are keyed must coincide with the locations of the designated water quality monitoring sites or ecological survey sites. This approach will facilitate analysis of chemical, physical, and biological parameters in relation to watershed-landscape level human activities. In addition, field verification of watershed boundaries will likely be required, especially in urbanizing watersheds with roads and stormwater conveyance networks. In typical urban watersheds the natural topography is often modified during the development process. Roads often block natural drainage paths and stormwater runoff is frequently routed outside its source drainage basin.

Wetland delineation may start with the National Wetlands Inventory (NWI) database, but should include regionally and locally delineated wetlands as well. Again, some fieldwork will be necessary to "ground-truth" wetland locations, extent, and characteristics. The U.S. Fish and Wildlife Service (USFWS) NWI is a 1:24,000 scale data set with wetland delineations and type attributes. These data were derived from photo interpretation techniques and often represent the best available coverage of the study area. The information will be used to provide a simple measure of wetland area per watershed area, which will be used as one of the "metrics" of watershed condition. In addition, any historical information about the location and characteristics of wetland areas that have been filled or drained is also of interest in both the hydrologic and ecological analysis of the watershed.

2. Soils & Topography—Mapping of soil types can usually be obtained from Natural Resource Conservation Service (NRCS) soil surveys. Knowledge of soil type distribution is useful from a stormwater perspective in that infiltration potential for various soil types is important in the selection of best management practices. The delineation of steep slopes and potentially unstable areas should also be incorporated into a GIS layer. This layer is useful in identifying potential problem areas for surface erosion and mass-wasting sites. Typically, slopes greater than 30 percent are considered to have a higher potential for mass-wasting events. Shallow landslides are perhaps the most common of these events. A combination of steep slope, non-cohesive soils, and topographic features that concentrate drainage often lead to landslides. Landslides are natural events, but can be exacerbated by

land-use activities (such as mining operations, timber harvest, road-building, construction, and development) that can concentrate surface drainage, saturate sub-surface soils, and/or remove natural vegetation that maintains slope stability by natural root strength. In addition to GIS analysis using a DEM, aerial photos should be analyzed for existing or historic slide locations, as well as indicators of potential future mass-wasting events. Based on this analysis, a landslide inventory can be developed, which can then be transferred to the GIS database as a separate layer or combined with the DEM steep-slope analysis layer to produce a mass-wasting "hazard" map. It is likely that field reconnaissance will also be necessary to fully characterize the study watershed. Having a geologist and a soil scientist as part of the multi-disciplinary watershed assessment team aids in this process.

Surface erosion is also a major concern in the watershed analysis process. Overland flow of stormwater is greatly attenuated, or even rarely occurs, under naturally forested conditions; because an absorbent, complex layer of organic material, vegetation roots, and groundcover, as well as the forest canopy itself, usually protects soil. In meadow or prairie ecosystems, soils are also rarely exposed to rainfall or runoff, and are similarly protected from erosion by natural vegetation. Areas where soil is exposed due to construction activities, roads, agriculture, or other human land-use activities are prone to surface erosion. Sediment from erosion that is transported to streams, lakes, or wetlands can cause significant ecological damage to benthic habitat. To minimize erosion problems, erosion and sediment control best management practices are commonly required in most jurisdictions. Aerial photos, land-use maps, and road maps should be used to identify potential surface erosion sites. Extensive field surveys should focus on construction sites, agricultural areas, and roads. These surveys should concentrate on areas in close proximity to receiving waters and areas where erosional sediments can be conveyed into the natural drainage system by roadside ditches or stormwater piping networks. Based on the information gained from this analysis, an erosion-potential and problem area GIS layer or map can be created.

3. Roads & Transportation—This GIS layer should include all public and private roads, as well as highway and railroad right-of-way areas. Ideally, roads should be identified by their surface composition (paved, gravel, dirt, etc.) and their size (i.e., single-lane, residential street, major arterial, multi-lane highway, interstate freeway). Roads may also be classified by average daily traffic (ADT) level. Based on existing knowledge of the importance of the transportation component in urbanizing watersheds, the analysis of this data set is a critical part of the overall watershed analysis. In a typical urbanized watershed, roads and other transportation-related areas (park-and-rides, parking lots, etc.) can account for upwards of 60 percent of the total impervious area. Therefore, it essential to have a complete and accurate picture of the watershed road network. For urbanizing watersheds especially,

the accuracy of the road-layer is as important as the stream layer. For example, if a stream flows closely parallel to a road and both are not located accurately in the GIS, it may appear that the stream and the road cross each other several times when in fact they do not.

In addition to utilizing road data for impervious surface area calculations, these data can also be used to calculate road density (km of roads per km^2 of watershed area), which tends to be closely correlated with imperviousness and has been widely used as a measure of human impact in areas where forest management is the dominant land-use. In urbanizing watersheds road-density is also an important land use parameter for quantifying human impacts. Finally, the road layer will be utilized in determining the stream-riparian corridor fragmentation, based on the number of stream crossings within a stream system (typically normalized by stream channel length). The greater the number of stream crossings, the more fragmented the stream-riparian ecosystem and the greater the opportunity for stormwater to enter the stream network. Road crossings also have the potential to become migration barriers for aquatic biota and can be "choke-points" in the urban drainage network, resulting in frequent localized flooding events. In this respect, stream crossings should also be classified as bridges or culverts. If possible, culverts should be further classified by type, size, and status as migration barriers for fish and other aquatic biota.

4. Stormwater Infrastructure – A GIS layer covering the stormwater piping network and the location of stormwater treatment facilities (BMPs) should be developed if not already available. Also useful are the locations of roadside ditches, swales, and other *de facto* stormwater conveyance or treatment devices. If possible, include details on stormwater piping such as size and condition. Each stormwater treatment BMP facility or device should be identified, located, and classified based on design specifications as well as operational and maintenance criteria. It is also desirable to delineate the drainage area and boundaries of individual stormwater conveyance-treatment sub-basins.

   In some cases, stormwater runoff could have an overriding influence on water quality. The stormwater system normally is composed of conveyance and treatment structures. These man-made structures may distort the relationships that are being sought between watershed or riparian buffer characteristics and water quality parameters. A riparian area could be in a good natural condition (e.g., mature forest) and still be associated with a degraded stream habitat when drainage pipes contribute stormwater directly to the stream from developed areas that are external to the buffer zone. The stormwater discharge from the developed area may have a elevated peak runoff rates and volumes, as well as transport high levels of contaminants to the stream. The metrics initially included in the GIS analysis protocol would not completely address this situation. Information on stormwater piping

density could play an important role, but the median distance from the nearest stormwater outfall may not accurately reflect the true influences on stream conditions. Another metric could be added if the stormwater network is available in the GIS. The best metric would reflect the actual discharge coming from external sources and provide a clear link to hydrology and water quality. The addition of outfall discharge could be accomplished for areas that have conducted detailed watershed analyses. If a watershed model has already been developed, GIS maps of the piping network and the estimated discharge or a given design storm should be available for each outfall. Simple modeling procedures could also be used as part of the GIS protocols for those jurisdictions that do not already have these data readily available. Integrating the GIS protocols with existing analyses will also aid in making a more effective end product for watershed management.

5. Land Use and Land Cover (LULC)—The extent of natural vegetation cover and developed areas should be determined using the best available technology. Options include aerial photographs, land-use and zoning maps, and remote-sensing data. Whichever data source is utilized, that data should be digitized as a GIS layer. The land use and land cover data may need to be purchased as part of the watershed assessment project. Data sets are usually collected in their native file types or projections and can come as either raster, vector or other image formats. File types may range from spreadsheets with coordinate pairs to Arc-View Shape-files and Arc-Info Grids. AutoCAD data and other formats can also be utilized if they are readily convertible into GIS layers. A variety of projections can be represented in the original data, which may need to be transformed into a common coordinate system, although newer versions of the GIS software typically can deal with compatibility issues automatically.

   The LandSat-TM-7 data set is currently the most recognized and widely used remote-sensing method for obtaining land cover data. This data set, which includes land use, land cover, and impervious surface information, is the cornerstone of the GIS analysis. LandSat-TM-7 data provide information on current vegetative land cover (including vegetation type and maturity), the extent and type of development, and riparian corridor conditions. The LandSat-TM-7 data set is typically derived from satellite imagery at a 30-meter resolution. These data are then classified into land use and land cover categories using standardized classification techniques and category types. Digital ortho-photos may also be used to ground-truth the remote-sensing data. Essentially, each “pixel” of LandSat-TM-7 “raw” data must be classified as a specific vegetation type or categorized as a designated land-use.

   The Anderson LULC classification scheme (USGS 1976) provides a standard framework of LULC classifications for use in GIS and remote-sensing applications. This classification protocol should be followed as much as possible. However, in many cases, LULC categories will depend on zoning

plans and other regional or local factors. Land cover classification will also depend on local or ecoregional conditions. Natural vegetative cover will vary based on climate, soils, topography, and other factors. Natural conditions in some ecoregions tend towards a forested landscape, dominated by specific species of trees, whereas others may have a more patchy landscape. In most cases there will be a mosaic of vegetation types, stand-ages, and native land covers throughout the watershed even without human influences. It is critical that this natural or "reference" condition be established for comparison purposes throughout the watershed assessment process. Typical LULC classifications include:

- Agricultural or cropland
- Grazing or pastureland
- Rural residential
- Low-density suburban residential
- Medium-density suburban residential
- High-density urban residential
- Multi-family residential
- Community/Institutional
- Recreational (sports fields, golf courses, etc.)
- Commercial
- Industrial
- Airport runways, hangers, and flight-lines
- Bare Soil
- Forest (coniferous, deciduous, mixed)
- Shrub/Scrub/Savanna
- Meadow/Grassland/Prairie
- Parkland or open-space areas
- Wetlands
- Lakes

In addition to mapping current land-use, it is also useful to have a GIS layer showing future or planned land-use patterns. This layer can be derived from zoning maps or data from a regional or local comprehensive or "master" plan. Knowing where development is currently planned can assist in analyzing potential impacts and making recommendations for avoiding future problems. This layer can be used to perform a "build-out" scenario analysis based on planned development and to develop optional zoning plans based in "what-if" scenario analysis. It is also useful to have a GIS layer showing jurisdictional boundaries and other development-related features of interest. Some regions have growth management policies that stipulate the delineation of "urban growth areas" where development is supposed to be concentrated in the hope of reducing urban "sprawl". If historic land use trends are an issue, analysis of past LULC data may also be relevant. This assessment may be especially important in urbanizing watersheds where development is

incremental and difficult to track spatially and over time, or when trend analysis is important.

Comprehensive Watershed Assessment GIS Protocols

The consensus process involving project representatives from the Puget Sound, Austin, and Montgomery County regions produced the following recommended protocol for GIS-based watershed assessments. The assessment consists of GIS-based analyses at three scales: watershed, riparian, and local. Maps in Appendix C illustrate the outcome of the process at each scale.

GIS Analysis of Watershed Conditions—

1. Using the GIS hydrologic layer and the DEM, delineate the watershed boundaries for the drainage area upstream of the study reaches or survey points of interest (see Figure 5). Create a "base map" of the watersheds of interest showing stream channels, wetlands, lakes, and major natural, topographic features. In addition, main roads, jurisdictional boundaries, and other development-related features might be included on this map.

2. The next step in the GIS watershed assessment process is to develop the land use and land cover (LULC) layers. In general, each pixel is classified into a particular LULC category, and groupings of the same type of LULC will form polygons or "patches" of LULC. If a LandSat-TM-7 dataset is being utilized for land and vegetative cover analysis, pixels and polygons of each land cover category are identified and quantified within the system. In addition to patch size and number of patches, the perimeter of each patch can be measured. Based on these data, a variety of indices of landscape fragmentation can be calculated (Farina 1998). This step in the GIS analysis process not only quantifies watershed land cover, but also classifies and quantifies each land use category by the same polygon analysis. It should be noted that the location of land use patches within the watershed, especially in relation to sensitive aquatic resources or habitat areas, is also important to the overall analysis.

   Similarly, patches of forest or other native vegetation (e.g., prairie, wetland, etc.) should also be delineated. In addition, "gaps" in the natural forest canopy should be delineated as polygons. As part of this phase of the watershed assessment process, the working scale of delineation should be set (1:4800 is suggested based on current experience), and a minimum patch or gap size should be established for the analysis (this setting likely will depend on the resolution of the data available). There should be specific "rules" established to guide the delineation of forest patches and gaps. For example, in urbanizing watersheds, roads are ubiquitous and may be the major cause of forest fragmentation. However, depending on the width and configuration of the road, a break in the forest canopy may or may not occur.

In general, roads adjacent to forest gaps should be included in the “gap” polygon. If possible, each forest patch should also be classified by the dominant type of forest comprising that patch (e.g., coniferous, deciduous, or mixed).

If feasible, forest canopy cover should be assessed using aerial photographs or other remote-sensing data. This characterization can provide a measure of forest quality or maturity. In general, more mature forests tend to have a more closed canopy structure. Various schemes could be developed to categorize forest canopy closure based on regional forest characteristics. In the Pacific Northwest, forest stand-age data have been classified based on LandSat-TM-7 (30-meter image resolution) and aerial photo analysis into crown closure classes. Also in this region, the combination of forest canopy closure and forest type can be used as a measure of hydrologic maturity of the forest or its ability to maintain the natural hydrologic regime of the watershed. In general, the more coniferous trees and the greater the canopy closure (maturity) of the forest, the greater is the interception and evapotranspiration.

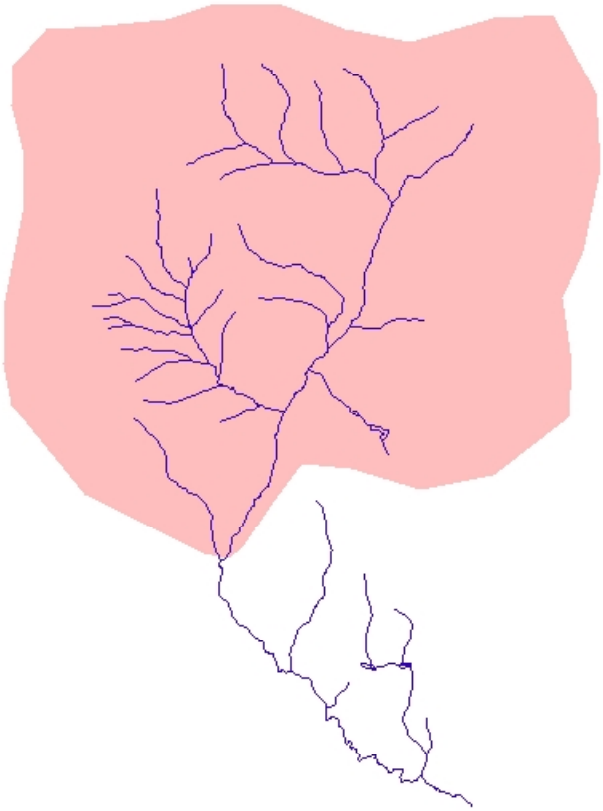


Figure 5. Watershed or Sub-Watershed Delineation

The end product of this portion of the analysis phase is an integrated LULC map or maps that will be the basis for further, detailed analysis of watershed conditions. These GIS layers may also be input to a watershed "model" that could be used to predict water quality on the basis of standardized values assigned to different land use or land cover categories. These models can be regionally developed or modified from existing models such as the U.S. EPA's "BASINS" model (http://www.epa.gov/ost/basins/).

Based on the results of the LULC process outlined above, the impervious surface area of each sub-watershed can then be determined. Directly establishing impervious cover on any significant scale is very labor-intensive. As an alternative, a regionally validated set of conversion factors should be incorporated into the GIS algorithm to calculate imperviousness based on land use and land cover. The relative location of impervious surfaces within the watershed is important as well, especially those impervious areas that are hydrologically connected to the stream system (effective impervious area or [EIA]). In most cases total impervious area (TIA) is an acceptable measure of development.

3. As an additional measure of watershed urbanization, road density should also be calculated using GIS. In addition, the total length of each type of road (highway, arterial, residential, etc.) should be tabulated on a watershed level. The fragmentation of the stream riparian corridor should also be measured by tallying the number of road crossings per unit length of stream channel. Current research indicates that frequent breaks in the riparian corridor can result in a measurable decline in ecological integrity (May et al., 1997a, b). If a GIS layer for stormwater infrastructure is available, the number of stormwater outfalls draining to the stream system should also be tallied and normalized by stream length, as was done for road-crossings.

4. The following is a partial list of potentially useful watershed-scale landscape metrics (see the Watershed Analysis Matrix in Appendix D for a more complete list):

   - % TIA and/or % EIA
   - % developed area
   - Road density
   - % forest cover
   - % wetland area
   - % natural land cover
   - Number forest patches per unit watershed area
   - Number forest gaps per unit watershed area
   - Average patch size
   - Average gap size
   - % urban land use

- % suburban land use
- % agriculture
- % rural land use
- % commercial land use
- % industrial land use

The analysis procedure outlined above was assumed to be for the entire watershed under study. The same analysis can also be done for any number of "nested" sub-watersheds within the drainage area under study. In this way, a sub-watershed analysis of the upper, middle, or lower watershed can be conducted, as well as analyses of individual tributary sub-basins. Using this analysis methodology, small catchments can be analyzed individually and then combined into a cumulative impacts analysis.

GIS Analysis of Riparian Conditions—

1. Use the GIS hydrologic layer to identify the stream(s) of interest. In many cases an entire stream sub-watershed may be under study, but in some instances a specific reach of the stream is the focus of study. Using the DEM, delineate the sub-watershed boundary for the catchment area upstream of the study reach or survey point. Delineate all natural stream channels (permanent or intermittent) upstream of the study reach. In the case of a full stream study, all stream channels and tributaries within the watershed should be included. In addition, identify and delineate any floodplain areas, off-channel wetlands, estuary deltas, or channel migration zones (CMZ). The final delineation of the stream system should include all of these hydrologic features in the GIS layer. This layer will serve as the foundation for the GIS analysis of the stream-riparian corridor.

2. Use GIS software to create a "buffer" of specified width on both sides of the stream system under study (see Figure 6). This "banding" process is a GIS-based analysis that is used to determine the composition of each buffer area with regard to the specified land cover and land use categories. Selection of banding increments should depend on what riparian buffer widths are currently required by local regulations and other widths that may be under consideration. It is also important to analyze for buffer widths that are ecologically meaningful for the ecoregion in which the study stream is located. Buffer widths of interest may include some combination of 1, 3, 5, 10, 15, 30, 50, 100, and 300 meters.

   The selection of buffer widths should be driven by scientific as well as policy considerations. In most cases there will also be a GIS limitation on the selection of the narrowest buffer width, based on the resolution of the data being used in the analysis.

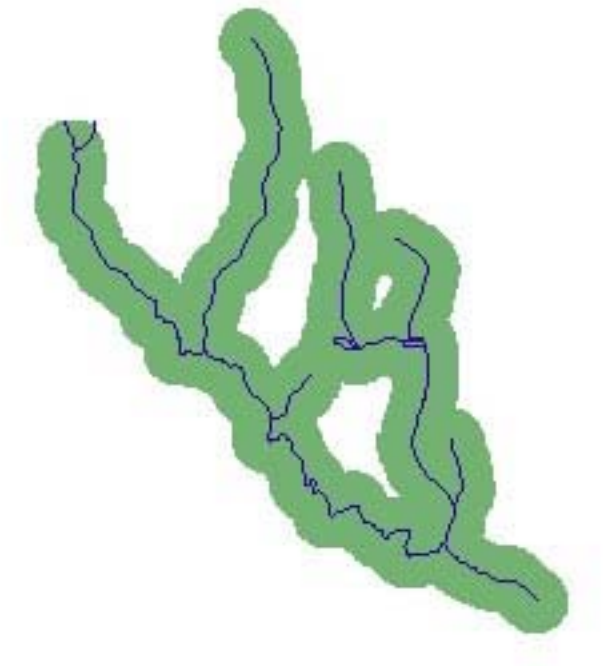


Figure 6. GIS Buffer for Riparian Scale Analysis

3. Overlay the land cover and land use GIS layers with the riparian buffer zones established above. Within each "band", analyze to determine the total area of each land cover or land use type within the buffer area. As a first increment of the riparian analysis process, analyze for only two "composite" categories of land cover; the natural vegetation areas and developed areas. The "natural" land cover category should include all native vegetation types for the specific eco region in which the study stream is located. Cultivated or grazed fields, lawns, turf areas such as sports fields or golf course, and landscaped areas should not be included in this category, unless it is determined that they provide full riparian structure and function for the eco region of the study. Any land cover or land use that does not meet this "natural" criterion should be considered "developed" and be grouped into that category. This initial "lumped" analysis will identify where human encroachment into the designated riparian buffer zone has occurred. This analysis should be performed for each buffer width of interest. It should be noted that this stage in the analysis process might be adequate to meet the objectives of the project.

   The next step in the "banding analysis" is to measure the riparian vegetation quantity and quality. To do so a more detailed analysis is necessary. As the next step in the riparian assessment process several broad categories of natural vegetation should be selected for banding analysis. If possible, forest canopy cover should also be determined as a measure of forest maturity. In many ecosystems, a mature forest often provides the optimal level of riparian

structure and function and is therefore the desired vegetative condition. In other ecoregions maturity is not as critical. The following land cover classifications are suggested for regions where a forest community dominates the riparian corridor:

Coniferous forest	Deciduous forest
Mixed forest	Wetlands
Shrub or scrub	Grassland or prairie

The listed categories of natural land cover are appropriate in temperate, humid regions, but other categories may apply in other ecoregions, especially arid or semi-arid locations. In the Puget Sound region, forest has been classified as coniferous-dominated (>70 percent of canopy), deciduous-dominated (> 70 percent of canopy), or mixed (both < 70 percent of canopy). Natural riparian conditions should be specifically defined for each study region based on regional "reference" (optimum natural) conditions. Ideally, an evaluation of forest maturity would also be desirable. If the data are available, categorize forest areas based on maturity. For example, forests could be delineated by stand-age or classified as old growth, mature, and young.

In some cases, the inclusion of marine, estuarine, and nearshore areas in the overall riparian assessment may be desired. For most regions of the country, much less is known about nearshore riparian reference conditions than freshwater systems, but treating these areas as equivalent to the freshwater areas is a reasonable initial step. The marine (or lake) shoreline can be analyzed much the same as the stream corridor, using the "buffer" and "banding" process.

As the next step in the process, it may be desirable to classify further developed land uses within each riparian buffer. In general, identification of developed and natural areas may be adequate for the initial assessment of riparian conditions. However, more detail on land cover and land use may be required for management, restoration, or conservation activities. Therefore, a detailed delineation of land-use activities in each buffer zone may be needed.

If possible, break down agricultural land-use into:

Active row-cropping	Active pasture or grazing

If possible, break down developed land use into categories such as:

Single-family residential	Multi-family residential
Rural residential	Suburban residential
Commercial	Industrial
Institutional	Recreational

Highway or roads	Airport
Turf or lawn	Open space or parkland
Golf course	Mining or gravel pit

4. Next, use the GIS system to count the number of breaks (crossings by roads, trails, pipelines, utilities, and other human intrusions) in the stream riparian corridor. Normalize this measure of *riparian fragmentation* by dividing by the total length of stream channel for the study stream or reach under analysis. Detailed identification of bridges and culverts should be included in the assessment process if feasible. This step will likely require some field verification as well. Another indicator of human intrusion into the riparian corridor is the presence of stormwater outfalls. In general, these features should also be evaluated based on the number of outfalls per unit length of stream channel.

5. Create an analysis table for each buffer width along with the land use and land cover categories used. In addition to the *%-developed* category and the *%-natural riparian* classification, other categories (such as those listed above) may be required to meet the study objectives. In addition to quantifying riparian buffer widths, this analysis will provide a measure of "riparian quality" and identify development encroachment. Encroachment can be defined as any natural riparian buffer that is less than a specified width on either side of the stream channel. Include riparian fragmentation data in the analysis table as well.

<u>GIS Analysis of Local Conditions</u>—

The combination of watershed or sub-watershed and riparian scale landscape analyses can explain much of the variability in ecological (physical, chemical, and biological) data. However, in some situations, there is a level of degradation that cannot be fully explained by the larger scale landscape metrics. In these cases, experience has indicated that local human activities or land use practices can have a significant impact on in-stream conditions and may help to explain why conditions are not what would be expected based on upstream sub-basin and riparian characteristics. Research in the Puget Sound region indicated that local conditions surrounding the water quality or biomonitoring sample site may be the cause (Booth et al., 2002). For this assessment, the following analysis was developed.

1. Use the GIS hydrologic layer to identify the stream(s) of interest. Identify and mark the site(s) where chemical water quality samples were obtained, where biological monitoring was performed, or where physical habitat measurements were taken.

2. Using GIS, delineate a *local influence zone* (LIZ) or an "arc of human influence" around that sampling site. It is important to analyze for a local influence zone that is meaningful from both an ecological and anthropogenic perspective for the

region in which the study stream is located. A LIZ can be circular or rectangular in shape. For example, it could be a circle of 10, 30, 50, 100, or 300 meters in diameter extending from the sampling site upstream. Alternatively, it could be a rectangle extending 100 meters on either side of the stream and 1 km upstream.

3. The purpose of this "local" analysis is to identify any significant localized contributions to the cumulative impacts measured at the sample site. Research has indicated that under certain circumstances, local influences can obscure or even overwhelm watershed and riparian scale effects on ecological or biological conditions. An example of this effect would be the use of toxic chemicals within a short distance of a sample site with little or no use of those same chemicals within the watershed as a whole.

4. Using GIS, analyze these LIZ areas for the same LULC metrics used for the watershed and riparian scale analyses described previously. Create a data table for these local-scale results.

Application in the Three Regions

The consensus system was applied in the three regions using the most complete available remotely sensed data set defining land use and land cover at or near the time(s) of stream biological and habitat assessments. For Puget Sound LANDSAT data were employed with the classification procedures developed for the region by Hill, Botsford, and Booth (2000). The classes of land cover produced were chosen to reflect the categories that can be readily distinguished in the satellite data and to have important differences in their associated runoff and watershed characteristics. Austin used a modified form of the classification system originally introduced by Anderson et al. (1976). Aerial photography in the best available resolution was the basis of LULC classification. More specifics are available on the City of Austin website http://www.ci.austin.tx.us/landuse/survey.htm. Montgomery County digitally compiled land cover information from aerial photographs and used it to develop GIS layers. For specifics refer to http://www.montgomerycountymd.gov/siteHead.asp?page=/mc/services/dep/index.html.

METHODS FOR COMPREHENSIVE DATA ANALYSIS

RECONCILING INDICES AMONG REGIONS FOR COMPARISON PURPOSES

All three regions represented in the study developed benthic macroinvertebrate indices based on community metrics appropriate for the regional ecology. In each case the index consisted of a summation of scores for the metrics. With differing numbers of metrics, the respective regional indices had different maximum and minimum values. To place all indices on a common base for comparison, each was expressed as "percent of best integrity" according to:

$$\text{\% of Best Integrity} = \left(\frac{Score - Minimum}{Maximum - Minimum}\right)x100$$

The same procedure was applied to the watershed condition developed as described below, as well as to the Montgomery County fish index of biotic integrity.

EXPLORATORY STATISTICAL ANALYSES

The classifications of pervious and impervious land cover made possible by the complete GIS databases produced in the first Phase 4 task provided much more information about conditions at the watershed, riparian, and local scales than previously available. Regional data analysis began with routine statistical explorations of possible associations between stream ecological variables and these various environmental attributes at the several scales. These examinations were performed with Statistical Program for the Social Sciences (SPSS) 10.1 for Windows software.

The first investigation was bivariate correlation analyses on the full matrix of landscape and ecological variables. The intent was to identify the landscape variables exhibiting the highest correlations with measures of habitat quality and the macroinvertebrate and fish communities to inform subsequent analyses.

The next exploration employed multiple linear regression techniques. In multiple regression analysis values of the dependent variable y (in this case, an ecological measure) are estimated from those of two or more independent variables x_i (here, landscape measures) using an equation derived from the data in the form $y = b_0 + b_1x_1 + b_2x_2 + \ldots + b_nx_n$, where the parameters b_i are the partial regression coefficients and the intercept b_0 is the regression constant. Equations were derived using strategies of both forced entry of selected variables into the equations and stepwise acceptance of variables from the full set of land cover attributes at the three scales, based on statistical criteria for entry. The suitability of the resulting multiple linear regression equations was judged on several grounds (Zar 1984):

- Adjusted R^2 (square of the multiple correlation coefficient), an unbiased estimator of variance of the dependent variable accounted for by the regression;

- Significance of the F-statistic generated in the regression analysis of variance (anova) testing the linear relationship between the dependent and independent variables;

- Significance of partial regression coefficients and constant based on t-tests;

- Standard error of the estimate of the value of the dependent variable using the equation; and

- Examination of a plot of regression standardized residuals versus regression standardized predicted values to check adherence to the underlying assumptions of the regression method of normal distribution and equality of variance of values of the dependent variable.

The third exploratory analysis involved the development of logistic regression equations. A logistic regression equation allows predicting the probability that an ecological measure is in a certain group (e.g., >75 percent of the maximum possible value) based on one or more independent variables (here, landscape measures). A logistic regression equation has the form (Everitt and Dunn 2001),

$$\ln\left(\frac{P}{(1-P)}\right) = b_0 + b_1x_1 + b_2x_2 + \ldots + b_nx_n$$

where P = the probability of a dependent variable observation falling in a certain group and ln signifies the natural logarithm. The term on the left side of the equation is often referred to as the logit of P, L. The predicted probability of being in the group, therefore, is, $e^L/(1 + e^L)$, where e = the base of the natural logarithms. Usually, predicted probability greater than 0.5 is taken as the basis for accepting the likelihood of membership in the group. Logistic regression equations were evaluated with respect to their ability to predict group membership and the significance of the regression coefficients for models with a single independent variable or the partial regression coefficients for multiple independent variable models.

CHARACTERIZING WATERSHED LAND COVER USING GIS DATA

Development of Regional Watershed Condition Indices (WCIs)

The utility of a numerical index incorporating land cover variables was explored for each region as an alternative to the multiple linear and logistic regressions equations developed as described above. Development of the indices followed a procedure analogous to that used by Fore, Karr, and Wisseman (1996) for the B-IBI and Horner et al. (2002) for the IRI. The steps in this procedure were:

1. Select the land cover variables at the watershed, riparian, and local scales shown by the bivariate correlation analysis to have the strongest associations

with biological measures. In practice, a number of variables in the range of seven to ten exhibited relatively high correlations; and then the correlation coefficients fell off markedly.

2. Review the list of relatively highly correlated variables and judge if any represent very similar attributes and might be dropped and any in the next tier of correlation coefficients represent different attributes that might be an asset to index development. Decide on the composition of variables that should be used for an initial trial WCI.

3. Plot each of the chosen land cover variables versus one or more biological variables. Here, the respective invertebrate community indices for the three regions were taken as the primary biological variables.

4. Assign ranges of the biological variable representing community integrity from relatively high to low in five steps. For this procedure, the assigned ranges were ≥90, 75-89, 60-74, 40-59, and <40 percent of maximum possible index value.

5. Visually determine what land cover variable range is consistent with each biological variable range, and assign scores to each range from 1 for the lowest interval to 5 for the highest. Often, the distinction was sharp; and there were no values of the biological variable outside of the identified land cover variable range. In some cases a small minority of points did not comply and would have some but not a large effect on the WCI.

6. Add the scores, divide by the maximum possible score, and express as a percentage of the maximum.

The Montgomery County data set consists of more than 460 stream reaches and was randomly divided into two subsets for initial WCI development and later independent verification of trends and models generated using the index. However, the full Austin and Puget Sound databases were used in the development process.

Statistically Examining and Finalizing the WCIs

The utility of the trial regional WCIs was examined by first plotting available biological variables against the index. The Austin data set has only an invertebrate community index, while the Montgomery County and Puget Sound databases also contain fish community measures (fish index of biotic integrity, F-IBI, and coho salmon:cutthroat trout ratio, CS/CT, respectively). Various models were investigated using Excel software to explain the relationship between biology and land cover as represented by the trial WCI (e.g., linear regression and variable transformations to assess logarithmic, power, and exponential regression fits). The adjusted R^2 was employed as the first screen of the models. Alternative models failed to improve on or increased only very

marginally the explanatory ability of the linear model, with only one exception. In that case both models were retained for additional examination.

The models carried forward were then assessed as described above under Exploratory Statistical Analyses, with three additions (also performed with SPSS):

- Plot of the frequency distribution of the regression standardized residuals, to examine if the residuals were adequately normally distributed to consider the model to be appropriate;

- Normal probability plot of the regression standardized residuals (expected cumulative probability versus observed cumulative probability), as an additional check on normality; and

- Plot of the regression standardized residuals versus values of the dependent variable (biological variable), to discern any pattern that might suggest that the fit might be improved with some adjustment of the metrics in the index.

With these evaluations of the regional WCIs complete, some trial adjustments were then made to see if the addition of one or more land cover variables would improve the model. Also, the deletion of one or more variables was attempted to see if improvement would occur, or if the model would be just as acceptable with a smaller number of metrics, and thus be less demanding of input data. The evaluations of these alternative indices were according to the same basis as outlined for the initial trial WCI.

Once the most appropriate models linking biological variables to the regional WCIs were identified, confidence limits for the estimates of the biological variables were computed (95, 90, and/or 80 percent limits, depending on the regional data set) using SPSS software.

Confirmatory Discriminant Function Analyses

As the final step in evaluation of Watershed Condition Indices, discriminant function analyses were performed using SPSS to see if using the WCI and its component variables independently would yield similar outcomes. If so, the similarity would be a sign of relative robustness in the WCI to place sites in their proper groups. Discriminant function analysis is a technique for combining independent variables (in this case, the land cover variables comprising the WCI) into a single new variable, the discriminant function, that best discriminates among values of the dependent variable according to a criterion based on the statistic Wilks' lambda (Everitt and Dunn 2001).

Discriminant function analysis (DFA) was performed as follows:

1. Classify biological data in ranges of biological integrity, i.e., points in the highest and lowest biological health groupings and the points falling between those groups.

2. Conduct DFA with the biological variable as dependent and WCI as a single independent variable.

3. Conduct DFA with the biological variable as dependent and each land cover variable making up the WCI as independent variables, with all of those variables forced into the analysis.

4. Conduct DFA with the biological variable as dependent and each land cover variable making up the WCI as independent variables, with those variables entered into the analysis in a stepwise fashion based on statistical acceptance criteria (the default Wilks' Lambda procedure of SPSS (Kinnear and Gray 2000) was used).

5. Compare the classifications resulting from the application of the logistic regression equations and the DFA exercises in steps 2-4. General similarity in results from these four different analytical approaches is a sign of robustness in the WCI metric.

6. For the analyses in steps 3 and 4, plot the first discriminant function score versus WCI to see if the relationship is close to linear. If so, the result is a sign that the development of WCI by a numerical indexing technique and DFA using WCI's component parts lead to similar outcomes, helping confirm the validity of both approaches.

CHARACTERIZING STREAM HABITAT

Researchers in Austin and Montgomery County employed the Rapid Bioassessment Protocols (RBP) of the USEPA (Plafkin et al. 1989). From their in-stream measurements they computed RBP indices, which were used to represent habitat quality in the data analysis reported here.

Puget Sound stream habitats were surveyed using a procedure originally developed by May (1996). The resulting data were used to develop a Habitat Quality Index (HQI) using the same procedure as outlined for the WCI. Parametrix, Inc. (2002) used a similar procedure to develop a habitat index of this type for Snohomish County, WA streams. The development started by performing bivariate correlation analyses on the full matrix of habitat variables, B-IBI, the coho salmon:cutthroat trout ratio, and WCI. The habitat variables most highly correlated with the biological and land cover measures were tentatively selected. The development then proceeded through the remaining five steps given earlier for the WCI and the same type of statistical examination to finalize the HQI.

ASSESSING LINKAGES BETWEEN WATERSHED AND STREAM ECOSYSTEM CONDITIONS

Graphical Analyses of Biological, Habitat, and WCI Metrics

The underlying general goal of the entire USEPA and WMI research effort is to be able to identify the types and degrees of watershed modification by urbanization that certainly or probably will lead to particular levels of aquatic ecological health. This goal was pursued in this culminating data analysis for the project in two contrasting ways: straightforward graphical assessments and development of probabilistic multivariate forecast models, the latter covered in the next section.

It was observed in earlier stages of the research that particular levels of relatively good biological health were never observed without various watershed and habitat conditions. These conditions were labeled “necessary but not sufficient,” because it was also seen that they appeared to permit the existence of relatively good health but did not guarantee it. A number of stream reaches always fell below the particular level of health even with the watershed or habitat condition fulfilled. Some other necessary but insufficient conditions were not satisfied, or some variable not measured was likely responsible. The converse was also observed: some watershed and habitat conditions inevitably were associated with particular levels of relatively poor biological health. These observations were instrumental in planning the graphical assessments.

In the first graphical exercise, scatter plots of biological and habitat indices versus WCI were examined to denote values of the WCI that must be met to achieve specified biological or habitat quality. Similar scatter plots related biological versus habitat metrics. Not all plots were amenable to clear identification of critical values of the independent variable. A sufficient number did yield sufficient information to provide a simple management tool, based on extensive regional data sets, for determining what is ecologically possible in the given landscape. The tool can be employed as one consideration in making land use decisions and planning for watershed preservation and stream enhancement and restoration projects.

In the next graphical assessment, confidence bands were plotted around the best model fits. Sites lying outside of those bands were noted and examined for characteristics that may explain the deviation. The explanation could also lead to management recommendations tailored to the site. For example, a site with a relatively high value of the habitat index but low-ranking biological health may be compromised by poor water quality, which could be ameliorated in management.

A part of this graphical exercise was to draw envelopes surrounding all or the great majority of the data points on the scatter plots. These envelopes define the range of observed ecological conditions along the land cover (WCI) or habitat index continuum. Therefore, they portray the ecological highs achievable, the lows that can occur, and the relative extent of site dispersion along that continuum.

One final step was taken to determine if the WCIs are an improvement over the combination variables representing impervious area, forest and wetland cover, and riparian integrity applied in the last phase of research and illustrated above in Figure 1. The same combination variables were computed for the complete data sets now available and used to develop linear and alternative models analogous to those based on WCI. The judgment criterion was a comparison of the highest R^2 values from those models with the R^2 statistics associated with the WCI-based models.

Development of Regional Logistic Regression Models for Probabilistic Forecast of Ecological Conditions

Model Development and Evaluation Procedure—

As seen in earlier phases of this work, the graphical presentations exhibited relatively wide dispersion of points, even using the multi-metric WCI instead of a single measure like TIA. The modest R^2 values associated with the regression models showed that they have limited ability to explain variance in the dependent variable, and hence to predict its value in any given condition. On the other hand, the first graphical exercise confirmed earlier observations suggesting that it is highly probable that the aquatic ecosystem cannot be relatively healthy without achievement of a certain land cover condition, and also that the probability is high that ecosystem health will be relatively poor with an opposite set of conditions surrounding the stream. These observations were formalized by developing logistic regression models with WCI as the independent variable. As pointed out earlier, such models compute the probability that a value of the dependent variable will lie in a certain range of values of the independent variable. For quantification purposes, they allow moving away from linear regression equations intended to forecast a specific value, although with a relatively large degree of uncertainty with typical environmental data.

For each of the three regions, logistic regression analyses were performed with the SPSS software to predict membership in groups representing the highest and lowest levels of biological health. Selections of what represents good and poor health differed somewhat among regions, depending mostly on the distributions of available data points and quality of the models resulting from the analyses. In other words a rigid definition (say, biological index ≥90 percent of maximum value) was not applied across the board if a different definition (say, biological index ≥80 percent of maximum value) would yield a better model, one more useable by the region in its management deliberations.

The logistic regression equations in terms of WCIs were, in general, developed and evaluated in the same way as in the initial exploratory investigation, although using just the one independent variable. Once the univariate equations were derived, there was some further examination if the addition of one or two unrepresented land cover variables would substantially improve predictive ability while still meeting statistical evaluation criteria. Confidence limits (95 percent) for the exponential of the regression

coefficient (e^{b_1}) (or partial regression coefficients for multivariate models) were estimated for the best equations.

In a similar fashion, development of logistic regression models was attempted to predict biological group membership in relation to habitat quality. The set of models together provide a quantitative portrait of the linkages between biology and land cover, habitat and land cover, and biology and habitat.

Significance and Application of the Models—

The logistic regression equations are used as indicated in the section on Exploratory Statistical Analyses above. The logit, L, is estimated according to $L = b_0 + b_1(WCI)$ [plus one or more additional terms if the model is multivariate]. Then, the probability, P, of membership in the group for which the equation was derived is computed as $P = e^L/(1 + e^L)$. Upper and lower bounds of the probability estimate, P_u and P_l, are calculated as $P_u = CL_{LU}/(1 + CL_{LU})$ and $P_l = CL_{Ll}/(1 + CL_{Ll})$, respectively, where CL_{Lu} and CL_{Ll} are the upper and lower 95 percent confidence limits for the exponential of the logit, respectively. Probabilities above 0.5 are taken as signifying likely group membership, and data points exhibiting lower probabilities are unlikely group members. The confidence bands permit exploring the likelihood of group membership of points in the marginal zone of acceptance just above and below P = 0.5.

Assembling the necessary land cover data to compute WCI (and any other variable added to the equation) then allows assessing the probable ecological status of an unmonitored stream site. This assessment could be applied to various kinds of questions, such as the likely effect of a land development proposal that would remove and alter natural cover or, conversely, a riparian zone restoration project. As another example, an expression relating biology and habitat could be used to predict the effect of habitat upgrading.

EXAMINATION OF THE EFFECT OF STRUCTURAL BMP PRESENCE

The detailed BMP information was not available in the other study areas to conduct a thorough examination of the influence of structural BMPs on stream ecosystems like the one performed in Puget Sound's Bear Creek watershed. The only broad-scale information was BMP coverage. For Austin and Montgomery County this was the percentage of the overall watershed served by structural BMPs. For Puget Sound (only some watersheds), it was the number of BMPs per unit watershed area, per unit developed area, per unit impervious area, and per developed and TIA percentage points.

To get some insight on whether or not BMP coverage might be instrumental in the ecological conditions, partial correlation analyses were performed with SPSS software to examine associations between ecological and BMP coverage measures with WCI

and then TIA as the controlled variables (Zar 1984). Controlling for these variables removes their influence and allows judging the effects of BMP coverage regardless of other instrumental watershed conditions. Partial correlation coefficients were examined for statistical significance as an indication of the influence of BMP coverage.

RESULTS AND DISCUSSION

EXPLORATORY STATISTICAL ANALYSES FOR THE THREE REGIONS

Bivariate Correlations

The associations among landscape and ecological variables were first explored by constructing bivariate correlation matrices for the three geographic regions. Table 5 lists the landscape variables most highly correlated with the available biological metrics. The Puget Sound salmonid fish metric exhibited the highest correlation coefficients (R) overall, followed by the Puget Sound B-IBI. The Austin and Montgomery County invertebrate metrics were less correlated with the respective landscape variable, and the Montgomery County F-IBI had the lowest R-values. In the two cases having both invertebrate and fish data, the same (Montgomery County) or mostly the same (Puget Sound) landscape variables were relatively highly correlated with both invertebrate and fish metrics.

Among all three regions the most prominent landscape variables relatively highly correlated to biological metrics were measures of total impervious area and forest cover at the watershed scale and in riparian buffer zones over a range of widths. Some other measures of urbanization also ranked highly, such as transportation and road land uses in Austin and Montgomery County, respectively. The Puget Sound GIS database had no equivalent measures, but urban landscaping (grass and shrubs) and urban landscaping in combination with paved land appeared in both the B-IBI and coho salmon:cutthroat trout ratio lists.

Multiple Linear Regressions

Exploratory stepwise multiple linear analyses were only modestly successful in generating relationships explaining relatively high proportions of the variance in the dependent biological variables and satisfying the assumptions underlying the regression method. The best relationship in terms of explaining variance was a model describing Puget Sound B-IBI as a function of: (1) watershed forest cover, (2) riparian canopy cover rating, (3) paved plus urban grass-shrub cover in a 300-meter diameter local zone upstream of the sampling location, and (4) watershed area (adjusted R^2 = 0.59). Adjusted R^2 values were lower for Austin (0.44) and Montgomery County (0.50). The landscape variables represented were: Austin—(1) transportation land use, (2) 10-meter riparian buffer zone residential land use, and (3) forest in a 100-meter diameter local zone upstream of the sampling location; Montgomery County—(1) watershed land cover by roofs, (2) landscaping, (3) wetlands, (4) pasture, and (5) roads and 10-meter riparian buffer zone forest.

The results of these analyses demonstrated only moderate ability to explain variance in, and therefore to predict, biological response to landscape conditions. They did signal that analytical ability would be likely to depend on a relatively small number of the numerous landscape variables available.

Table 5. Landscape Variables Most Highly Correlated with Biological Metrics for Three Regions

	Puget Sound		Austin	Montgomery County	
	Benthic Index of Biotic Integrity	Coho Salmon:Cutthroat Trout Ratio	Invertebrate Score	Benthic Index of Biotic Integrity	Fish Index of Biotic Integrity
Landscape variables[a]	Watershed forest[b] Watershed TIA[b] 300-m buffer forest[b] 100-m buffer TIA[b] 300-m buffer TIA[b] 50-m buffer TIA[b] Watershed paved + urban grass-shrub 100-m buffer forest[b] Watershed urban grass-shrub 50-m buffer forest[b] 100-m buffer natural grass-shrub-crop 300-m buffer natural grass-shrub-crop	Watershed TIA[b] Watershed urban grass-shrub 300-m buffer urban grass-shrub Watershed paved + urban grass-shrub Watershed forest[b] 300-m buffer TIA[b] 300-m buffer paved + urban grass-shrub Road density 100-m buffer urban grass-shrub 300-m buffer natural grass-shrub-crop 100-m buffer paved + urban grass-shrub	Watershed transport 100-m buffer transport 50-m buffer transport 100-m buffer TIA[b] 10-m buffer TIA[b] Watershed commercial 50-m buffer residential Local natural land cover[b] 30-m buffer TIA[b] Local TIA[b] 10-m buffer residential Stream road crossings	Watershed roofs Watershed roads Watershed parking Watershed TIA[b] Watershed cropland Watershed native forest Watershed pasture Watershed BMP density 50-m buffer native forest 100-m buffer native forest	Watershed roads Watershed roofs Watershed native forest Watershed TIA[b] Watershed parking 50-m buffer native forest 100-m buffer native forest
Correlation coefficient range	0.64-0.72	0.72-0.78	0.47-0.54	0.22-0.70	0.21-0.46

[a] The 10 variables exhibiting the highest correlations are listed in order, although the list is longer in some cases because of ties for the tenth position and shorter in one instance because only eight variables exhibited correlation coefficients with a significance level $P < 0.01$.

[b] Forest--≥86% of pixels in forest cover; TIA—total impervious area; transport—any transportation land use; local—100 m on each side of the stream extending 1 km upstream from the sampling point.

Logistic Regressions

Once again the Puget Sound data produced the most successful logistic regression analyses. A model based on, (1) riparian zone fragmentation, (2) riparian canopy cover rating, (3) watershed forest cover, and (4) paved plus urban grass-shrub cover in a 300-meter diameter local zone upstream of the sampling location was able to predict correctly whether or not B-IBI was > 75 percent of the maximum value in 92 percent of cases when it was not and 69 percent when it was, with three of the four variables statistically significant at $P < 0.10$. Another model in terms of urban tree cover plus variables 2 and 4 in the first model predicted correctly whether or not B-IBI was < 45 percent of the maximum value in 93 percent of cases when it was not and 65 percent when it was, with all but one of the variables statistically significant at $P < 0.05$.

The Austin and Montgomery County data sets yielded only one logistic regression model exhibiting relatively high predictive ability. An Austin model predicted invertebrate score > 75 percent of maximum correctly 97 percent of the time when it was not and in 60 percent of the cases when it was in this group. The model was based on: (1) stream road crossings (number/km) and (2) natural vegetation in a local zone extending 100 meters on each side of the stream for 1 km upstream from the sampling.

It was decided to proceed to additional analyses, since these exploratory attempts did not produce a unified portrait that would be broadly useful in the three regions to assess and manage aquatic ecosystems in urban areas. These analyses were useful in pointing out variable types that might be brought into more analyses, in addition to the relatively high correlates. For example, Puget Sound riparian fragmentation and canopy cover and urban land cover in a 300-meter diameter local zone were not among the most highly correlated variables, but they were selected in stepwise multiple linear and logistic regression analyses.

The logistic regression exercises indicated that it is easier to forecast if a biological condition <u>will not</u> fall into a particular group near the high or low end of biological integrity than if it will. This observation was seen in earlier graphical assessments and, as discussed in following sections, was verified in subsequent graphical and numerical analyses.

REGIONAL WATERSHED CONDITION INDICES (WCI)

Initial WCI Development

Development of a WCI for Puget Sound began with the selection of nine possible metrics. The majority were chosen because of their relatively high correlation with B-IBI and their representation of urbanization (TIA) and buffering (forest cover) in the watershed as a whole and the riparian zones relatively near (within 50 meters) and distant (up to 300 meters) from the stream. For initial trials three additional variables were selected to represent transportation land use (road density, km/km^2), riparian fragmentation (breaks/km), and urbanization in a 300-meter diameter local zone

upstream of the sampling location (as paved plus urban grass-shrub cover). Exploratory analyses indicated the potential utility of these metrics. They were then scored as described in the methods section above.

Austin and Montgomery County WCI development started with, respectively, nine and five metrics chosen using similar considerations as applied to the Puget Sound data. For Austin these trial selections were: TIA in the overall watershed and 10- and 100-meter riparian zones; transportation land use in the watershed and the 100-meter riparian corridor; TIA and natural land cover in a local zone 100 meters on each side of the stream extending 1 km upstream from the sampling point; commercial land use in the watershed; and stream road crossings. The initial Montgomery County choices were watershed TIA and land cover by roads, roofs, parking, and native forest.

Statistical Examination and Finalizing WCIs

The watershed condition indices derived for each region were examined by plotting biological indices versus WCI and investigating different model fits (e.g., linear, exponential, power). For Puget Sound, B-IBI fit best as a linear function of WCI with R^2 = 0.53. Some unrepresented variables were added to the WCI to see if a better fit could be achieved, but without success. To investigate a more economical WCI, in terms of the number of metrics incorporated, those variables relatively less well correlated with B-IBI were removed one by one and in combinations. The R^2 value declined somewhat, except when riparian fragmentation and road density were taken out, when it increased very slightly. The index was then tentatively finalized with seven metrics, excluding those two.

This tentative conclusion was evaluated further by performing the same analysis using the coho salmon:cutthroat ratio. Removing riparian fragmentation and road density yielded the same result as in the B-IBI analysis, thus confirming that a seven-variable index optimizes analytical ability versus data demands. The best fit for the CS:CT ratio was exponential with R^2 = 0.75. A power model actually gave a higher coefficient (0.81) but inferior statistics otherwise.

The Austin and Montgomery County WCIs were treated in the same fashion. Regressing the Austin invertebrate score on the initial WCI yielded R^2 = 0.34 for a linear fit. That coefficient could not be improved with variable additions and was undiminished by removing commercial land cover, road crossings, and TIA in the local zone. The Montgomery B-IBI versus WCI fit (linear, R^2 = 0.43) could be improved neither by adding nor removing metrics. The fish IBI fit in that case was much poorer, with an R^2 of 0.17. Table 6 lists the final composition of the watershed condition indices for the three regions. Refer to Appendix E for WCI scoring for all three regions.

There are some common elements in the WCI metrics for the three regions. Total impervious area and components making up impervious land use (e.g., automotive-related land covers, roofs) predominated in the selections watershed-wide and over a range of buffer scales. This dominance points out the importance of obtaining good

measures of impervious land cover in performing watershed analyses. Forest cover was also prominent. A local-scale metric was less instrumental but was useful to improve the representation of watershed conditions in the two cases where it was available.

Table 6. Metrics Incorporated in Watershed Condition Indices for Three Regions

PUGET SOUND[a]	AUSTIN[b]	MONTGOMERY COUNTY[c]
Watershed forest Watershed TIA 300-m buffer TIA 300-m buffer forest 50-m buffer TIA 50-m buffer forest 300-m local paved + urban grass-shrub	100-m buffer transport Watershed TIA Watershed transport 10-m buffer TIA 100-m buffer TIA Local natural land cover	Watershed roads Watershed roofs Watershed TIA Watershed parking Watershed native forest

[a] Forest--≥ 86% of pixels in forest cover; TIA—total impervious area; local—300-meter diameter zone upstream of the sampling location.
[b] TIA—total impervious area; transport—any transportation land use; local—100 m on each side of the stream extending 1 km upstream from the sampling point.
[c] TIA—total impervious area.

Discriminant Function Analyses

Discriminant function analyses (DFA) were employed to compare the classification of stream sites in terms of biological health using the Watershed Condition Indices versus performing the same classification with the individual component variables making up the WCIs. The individual variables were treated in two different ways: including all in the DFA and entering them into the analysis in a stepwise fashion based on statistical acceptance criteria. Similarity in classification would demonstrate the relative robustness in the indices as formulated to place sites in their proper groups. Table 7 shows the comparative results for the three regions.

Table 7 shows that sites in the highest and lowest integrity groups were generally more successfully classified by all three methods than those in the intermediate categories. There was no consistent pattern indicating that using the variables individually by either method either improved or diminished classification accuracy. Therefore, the WCI formulations appear to provide valid means of characterizing watershed conditions and conducting analyses involving the biology of streams in these catchments. The aggregate formulation offers the advantage of being easier to use in numerical and statistical analyses than a host of variables.

It is notable that the stepwise method of entering variables resulted in the acceptance of only three of seven of the Puget Sound variables (watershed forest, 50-meter riparian buffer TIA, and 300-m local paved + urban grass-shrub) and just one variable each in

the other two regions (watershed transport for Austin and watershed roads for Montgomery County). Classifications involving a limited subset of the variables neither consistently increased nor reduced classification ability. Therefore, grouping exercises like this could be performed using just these variables.

Plots of the first discriminant function scores for multiple-variable DFAs versus WCI produced relationships close to linear in all cases. This result is a sign that the development of WCI by a numerical indexing technique and DFA using WCI's component parts lead to similar outcomes, helping confirm the validity of both approaches.

Table 7. Results of Discriminant Function Analyses Independently Classifying Invertebrate Community Groups in Three Ways (Percentages of Sites Correctly Placed in Their Groups)

		INVERTEBRATE COMMUNITY INTEGRITY				
REGION	METHOD[a]	≤ 25% OF BEST	26-44% OF BEST	45-74% OF BEST	75-84% OF BEST	≥ 85% OF BEST
Puget Sound	WCI	65	37	41	33	69
	All WCI variables	61	47	38	67	85
	Stepwise	57	37	43	56	77
Austin	WCI	78	27	20	80[b]	-
	All WCI variables	78	46	7	100[b]	-
	Stepwise	67	36	20	80[b]	-
Montgomery County	WCI	69	19	20	45	40
	All WCI variables	60	31	8	51	31
	Stepwise	66	25	22	20	69

[a] WCI—classification based on WCI alone; All WCI variables—classification based on forcing all WCI component variables into the analysis; Stepwise—classification based on stepwise acceptance of WCI component variables on statistical criteria
[b] ≥ 75% of best for Austin because of few points available

HABITAT CHARACTERIZATION

Habitat was characterized in Austin and Montgomery County using the Rapid Bioassessment Protocols (RBP) of the USEPA (Plafkin et al. 1989). For Puget Sound a Habitat Quality Index (HQI) was derived as described in the Methods for Comprehensive Data Analysis. Development began with the selection of eight possible metrics, generally because of their relatively high correlation with B-IBI and their

representation of a variety of habitat attributes. The index was examined by plotting biological indices versus HQI. Some unrepresented variables were added to the HQI to see if a better fit could be achieved, but without success. To investigate a more economical WCI, in terms of the number of metrics incorporated, one variable relatively less well correlated with B-IBI was removed without any negative effect on the fit. The index was then tentatively finalized with seven metrics. This tentative conclusion was evaluated further by performing the same analysis using the coho salmon:cutthroat ratio, which confirmed that a seven-variable index optimizes analytical ability versus data demands. Refer to Appendix E for HQI scoring. The variables incorporated are: (1) glide habitat (as a percentage of total habitat), (2) substrate embeddedness (as a percentage), (3) pool frequency (number/km), (4) large woody debris frequency (as number of pieces/km), (5) large woody debris frequency (as number of pieces/bank-full width), (6) large woody debris volume density (m^3/km), and (7) stream-bank stability rating. Their correlation coefficients with B-IBI ranged from 0.38 to 0.56 and with coho salmon:cutthroat trout ratio from 0.44 to 0.77.

REGIONAL LINKAGES BETWEEN WATERSHED AND STREAM ECOSYSTEM CONDITIONS

Graphical Analyses of Biological Metrics Relative to WCIs

Figure 7 portrays B-IBI plotted against WCIs for the Puget Sound region. To allow inter-regional comparisons, both biological indices and WCIs are expressed as percentages of the "best" possible values, as outlined earlier in the Methods for Comprehensive Data Analysis. Letters on the graphs (e.g., **A**, **B**) denote watershed conditions generally necessary to reach certain levels of biological integrity. In the Puget Sound region achieving B-IBI ≥ 85 percent of maximum integrity requires that WCI be at least 75 percent of the best value (**B** on Figure 7), with most of the highest B-IBI scores lying above a WCI of 90 (**A**). While these watershed conditions are generally necessary for good biological health, they are not sufficient alone, as demonstrated by the numerous points representing lower biological integrity at relatively high WCI values. The land cover data collected in this work do not allow exploring the many potential reasons for the failures to achieve good biological conditions when watersheds are not heavily developed. Nevertheless, this analysis identifies the key watershed conditions that must be provided if there is to be any chance of meeting relatively high biological goals.

Point **D** on Figure 7 indicates that the B-IBI was inevitably below 50 percent of the best if WCI fell beneath 35 percent, and always dropped again to under 30 percent with WCI less than 20 percent (**E**). Therefore, while poor biology can occur even with moderate or even little disturbance by urbanization, this outcome is invariable with heavy levels of disturbance.

Figure 8 presents an analogous plot for the Puget Sound coho salmon:cutthroat trout ratio. With less points, the necessary watershed conditions are not as sharply defined; but the trends are similar as those evidenced by the B-IBI plot. Ratios above 3 and 2 required WCIs greater than approximately 70 and 50 percent, respectively. Once again,

these watershed conditions are necessary but not sufficient to achieve the given levels of the fish metric. Dominance by the more tolerant cutthroat trout was inevitable with WCI < 35 percent of the best condition and became overwhelming with WCI in the vicinity of 20 percent and less.

Figures 9-11 show the relationships between the biological indices and WCIs for the other two regions. The trends are similar to those noted for Puget Sound although less sharply defined, mirroring the lower correlation and regression coefficients in these data seen in statistical examinations. The highest levels of health in the invertebrate communities were generally seen only when WCIs were > 80 and ≥ 70 percent of best scores for Austin and Montgomery County, respectively. The lowest levels always occurred with WCI < 25 percent in both cases. The Montgomery County fish index of biotic integrity exhibited the weakest trends, but F-IBI could exceed 90 percent only if WCI was at least 70 percent of the best condition and always fell below 60 percent if WCI was under 25 percent.

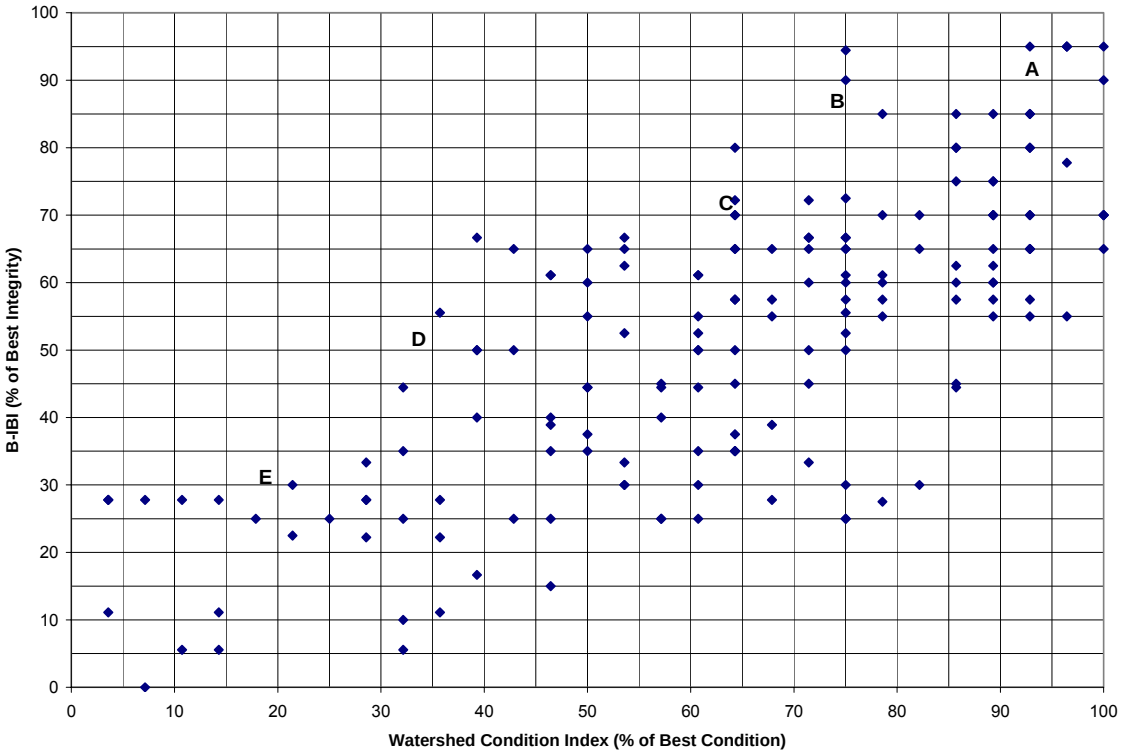


Figure 7. Benthic Index of Biotic Integrity in Relation to Puget Sound Watershed Condition Index [*Note:* **A**, **B**, **C**, **D**, *and* **E** *represent WCIs generally associated with B-IBI > 90, ≥ 85, ≥ 70, < 50, and < 30 percent of best integrity, respectively.*]

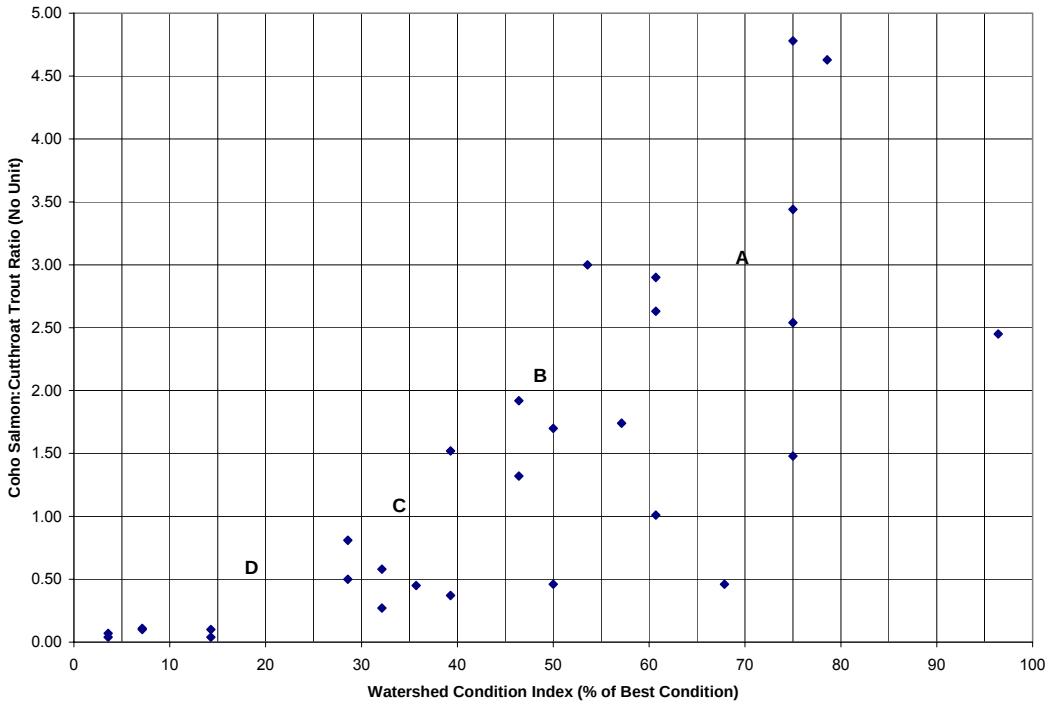


Figure 8. Coho Salmon:Cutthroat Trout Ratio in Relation to Puget Sound Watershed Condition Index [*Note:* **A**, **B**, **C**, *and* **D**, *represent WCIs generally associated with CS:CT > 3.0, > 2.0, < 1.0, and << 0.5, respectively.*]

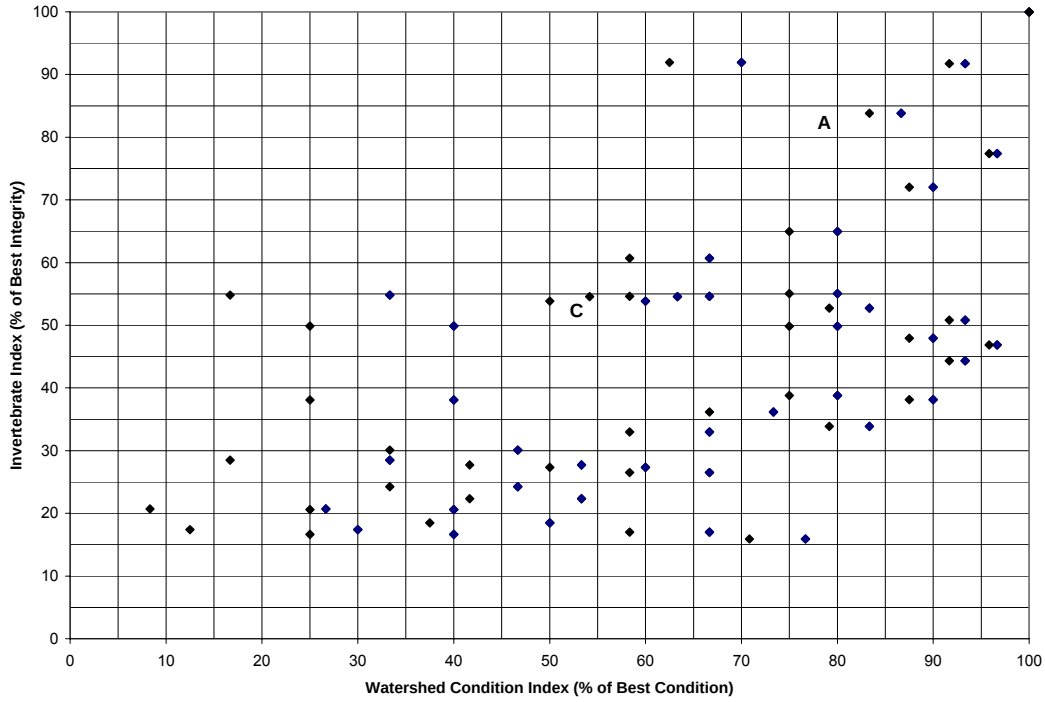


Figure 9. Invertebrate Index in Relation to Austin Watershed Condition Index [*Note:* **A**, **B**, *and* **C** *represent WCIs generally associated with Invertebrate Index > 80, > 55, and < 40 percent of best integrity, respectively.*]

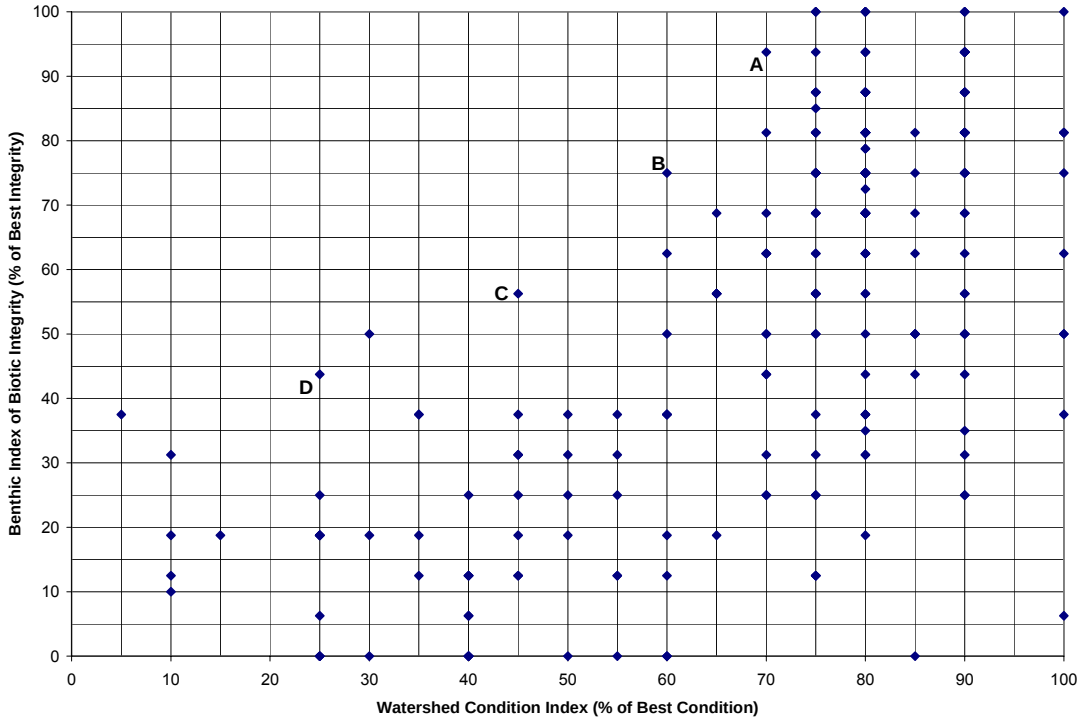


Figure 10. Benthic Index of Biotic Integrity in Relation to Montgomery County Watershed Condition Index [*Note:* **A**, **B**, **C**, *and* **D** *represent WCIs generally associated with B-IBI > 90, ≥ 75, ≥ 50, and < 40 percent of best integrity, respectively.*]

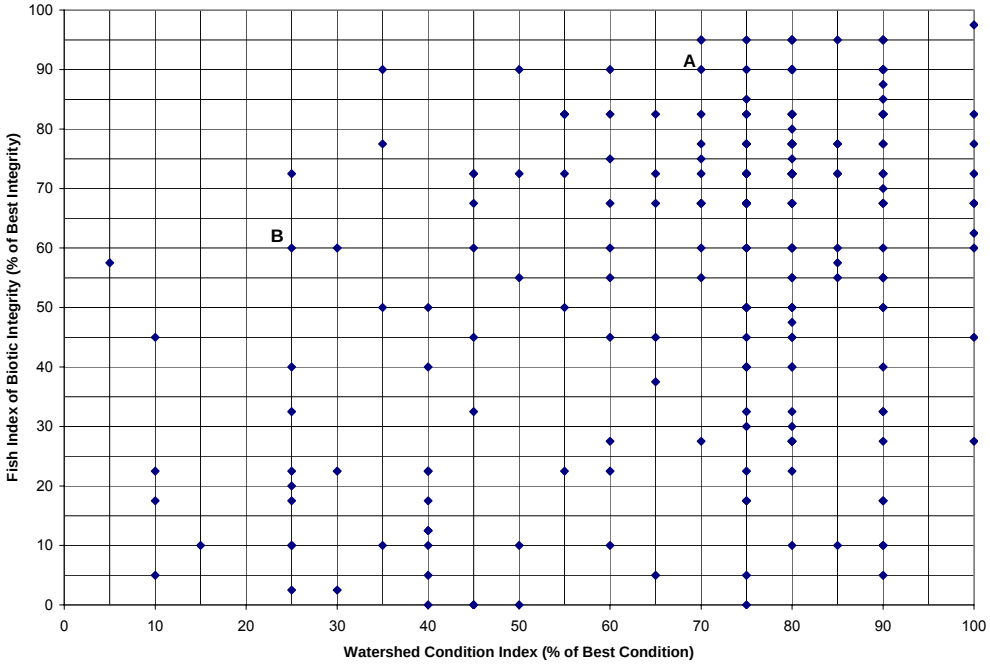


Figure 11. Fish Index of Biotic Integrity in Relation to Montgomery County Watershed Condition Index [*Note:* **A**, *and* **B** *represent WCIs generally associated with F-IBI > 90, and < 60 percent of best integrity, respectively.*]

The relationships between biological and watershed conditions were further examined by plotting the invertebrate indices linearly regressed on WCI and the confidence limits surrounding the regression line. Figure 12 provides this plot for Puget Sound. WCI explains just over half of the variance in B-IBI, and the regression has a standard error relatively high compared to the magnitude of the dependant variable. It is not intended to be a means of predicting B-IBI. As observed before, WCI represents necessary but not sufficient conditions to account for biological integrity; and it is thus not appropriate to use it alone as a predictor of specific values of a biological metric.

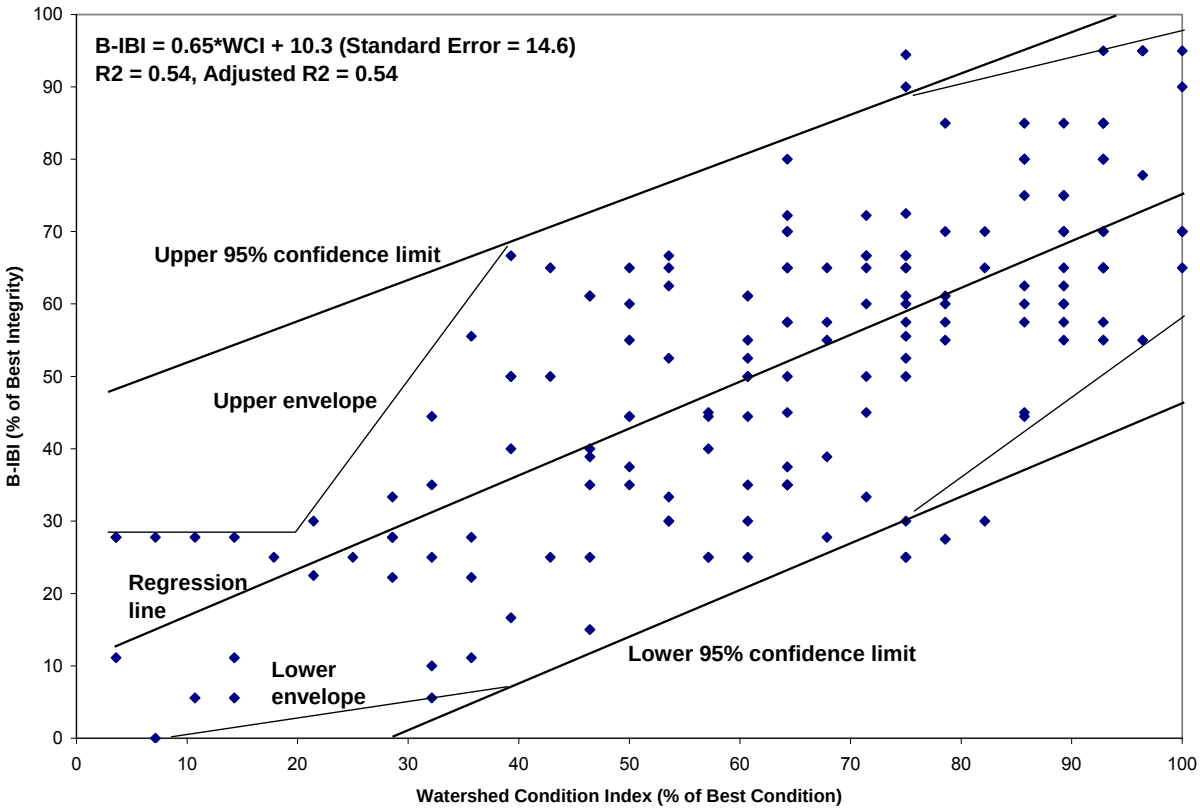


Figure 12. Linear Regression of Puget Sound Benthic Index of Biological Integrity on Watershed Condition Index with Confidence Bands

It can be noted in Figure 12 that the 95 percent confidence bands generally encompass well the spread of points between WCI values of approximately 40 and 75 percent of the best condition. Above and below that range, point dispersions narrow, more at the low end of biological integrity than at the high end. In the latter case this convergence reflects the general lack of negative influences on the biological community when urbanization, represented here by seven land cover metrics, is relatively light. The greater degree of convergence at the low end gives another indication of the inevitability of greatly reduced biological health with extensive alteration of natural landscape characteristics. Envelope lines were drawn by eye on the graph to illustrate the reduced point dispersions near the left and right sides.

Five points lie outside of the 95 percent confidence bands, two above and three below. Examination of the circumstances surrounding these sites can shed light on factors other than those directly represented by the WCI and can help in targeting watershed management. All three points with lower-than-expected biological integrity have watersheds with extensive hobby farm presence, a land use not represented in the WCI or, in any direct manner, in the underlying land cover database. Management would have to deal with these activities to improve biological conditions. It would be beneficial to bring this land use into future GIS analyses too. The site associated with the point just above the upper confidence limit has extensive, high quality headwater wetlands, mostly mature coniferous growth in riparian zones, and relatively new BMPs in upstream developments. Retaining biological health would rely on preserving the natural landscape features and maintaining the BMPs in good working condition. The other site in this zone has extensive riparian wetlands, a factor not explicitly taken into account in the WCI or the overall GIS database.

Figures 13 and 14 present the analogous plots for the other two regions. In both cases the linear regressions had lower R^2 values and higher standard errors than the Puget Sound regression. The Austin (Figure 13) 80 percent confidence band defines the locus of most points for WCI ranging from roughly 50 to 90 percent of best condition. As in Puget Sound there was a marked convergence of points at relatively low biological integrity, illustrated with the envelope curves. However, the convergence was less apparent at the upper end, where only one envelope line could be drawn. The 95 percent confidence band encloses all but two points, both falling above the upper limit. The Montgomery County (Figure 14) data do not exhibit the convergence seen at relatively high and low biological indices in Puget Sound and Austin. In this case the upper 95 percent confidence band embraces all points, although two fall well below the lower limit.

Finally, the utility of WCIs was investigated by comparing the linear models generated using these indices with equivalent regressions developed using the combination variables representing impervious area, forest and wetland cover, and riparian integrity applied in the last phase of research and illustrated above in Figure 12. For all biological measures in all three geographic regions, the R^2 values associated with those alternative models were lower than the R^2 statistics associated with the WCI-based models. Therefore, development of the watershed condition indices improves analytical ability overall in comparison to the preceding technique, and a substantial improvement over the common reliance on the single independent variable total impervious area.

Analyses of Habitat Metrics Relative to Watershed Conditions and Biology

Table 8 exhibits the adjusted R^2 values for linear regressions relating habitat with watershed and biological metrics for the three regions. It can be seen that poorer model fits resulted than documented above for most regressions of biological measures on WCIs. The exceptions were Puget Sound coho salmon:cutthroat trout ratio versus HQI and Austin invertebrate score versus RBP index. The relatively poor fits signify

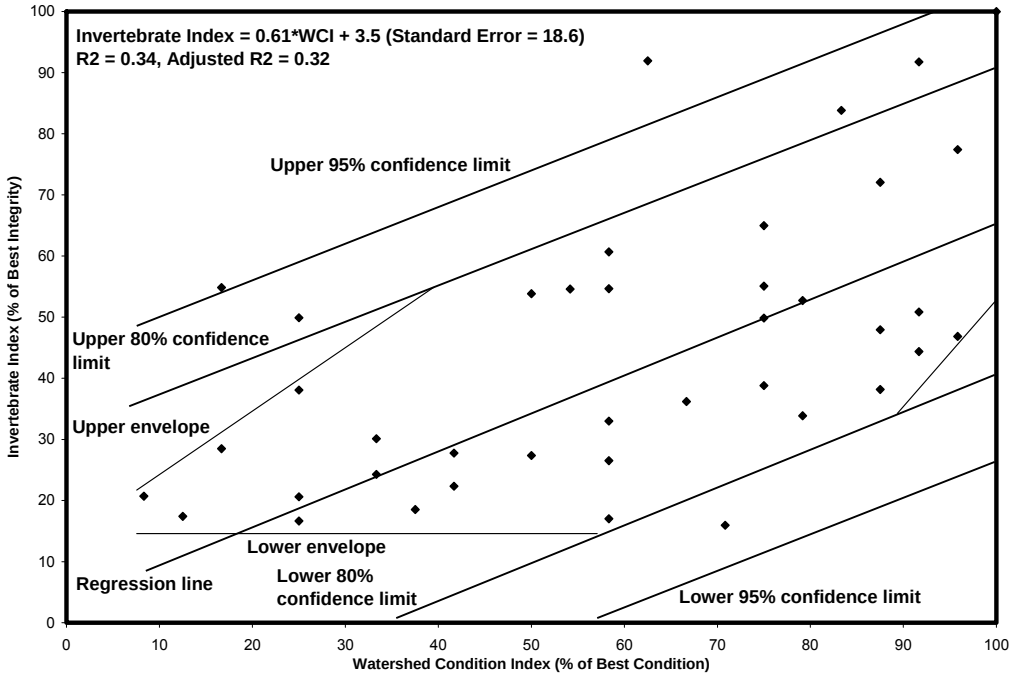


Figure 13. Linear Regression of Austin Benthic Index of Biological Integrity on Watershed Condition Index with Confidence Bands

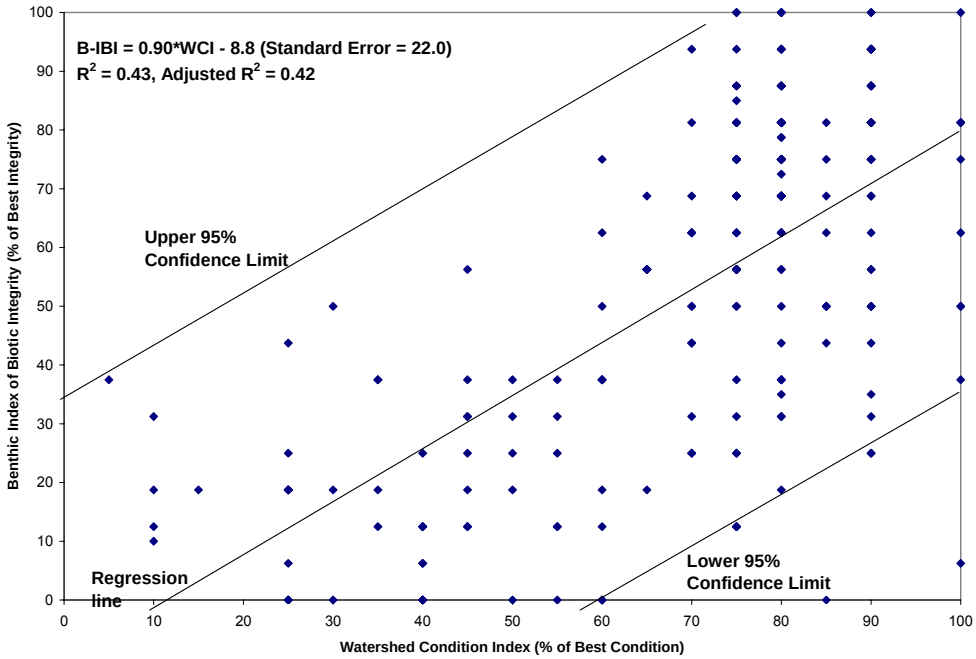


Figure 14. Linear Regression of Montgomery County Benthic Index of Biological Integrity on Watershed Condition Index with Confidence Bands

substantial scatter in the data and associated limitations in the ability to define with clarity regions on the graphs such as depicted on Figures 7-11 representing necessary conditions for given biological communities. Therefore, this type of analysis is not shown for most of the associations in Table 8. Figure 15 shows the best association in this group, for Puget Sound coho salmon:cutthroat trout ratio versus HQI. The two highest fish ratios (point **A**) both have HQI of 24 (maximum 28), but note that a habitat score at least this high does not guarantee an equivalent biological condition. Achieving CS:CT at least 2.0 (point **B**) requires HQI > 12, and below that point cutthroat trout dominance is almost inevitable (only one point excepted).

The generally relatively poor relationships between habitat and watershed conditions on the one hand and biology and habitat on the other encourages using biological response instead of habitat as a direct basis for making watershed management decisions. It appears that the many potential variables besides those represented in this analysis compound the problem of data scatter when attempting to describe relationships through the watershed-habitat-biology chain.

Table 8. Adjusted R^2 Values for Linear Regressions Relating Habitat with Watershed and Biological Metrics

LOCATION	LINEAR REGRESSION	ADJUSTED R^2
Puget Sound	Benthic Index of Biotic Integrity versus Habitat Quality Index	0.18
	Coho salmon:cutthroat trout ratio versus Habitat Quality Index	0.62
	Habitat Quality Index versus Watershed Condition Index	0.18
Austin	Invertebrate Score versus Rapid Bioassessment Protocol Index	0.08
	Rapid Bioassessment Protocol Index versus Watershed Condition Index	0.53
Montgomery County	Benthic Index of Biotic Integrity versus Rapid Bioassessment Protocol Index	0.23
	Fish Index of Biotic Integrity versus Rapid Bioassessment Protocol Index	0.08
	Rapid Bioassessment Protocol Index versus Watershed Condition Index	0.12

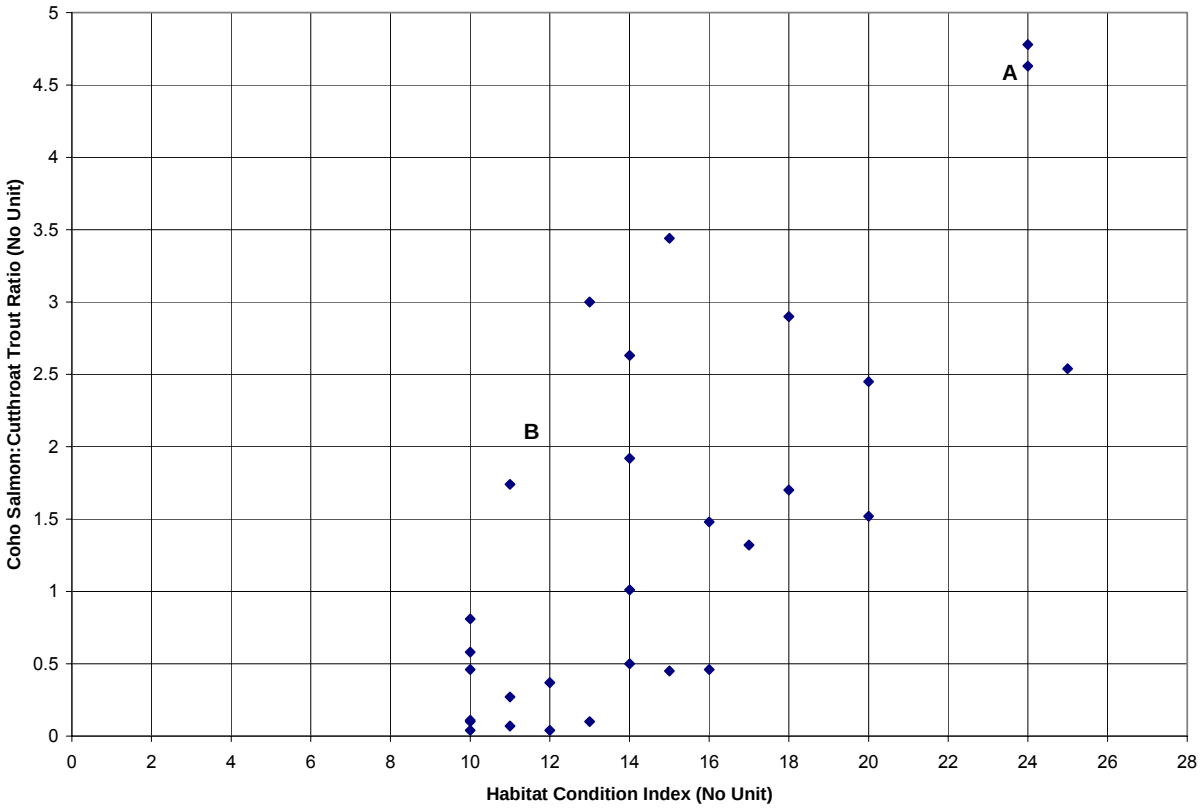


Figure 15. Coho Salmon:Cutthroat Trout Ratio in Relation to Puget Sound Habitat Quality Index [*Note:* **A** *represents WCI generally associated with CS:CT > 4.5;* **B** *represents CS:CT > 2.0 with WCI > its value at* **B***, and CS:CT < 1.0 with WCI < its value at* **B** .]

Regional Logistic Regression Models

Logistic regression equations were derived to forecast the probability of a stream's invertebrate or fish community being in selected groupings of biological integrity based on watershed conditions, principally represented by the WCI. Table 9 summarizes the results. Regression coefficients were statistically significant for all but one of the 16 models, the majority at the highly significant < 0.001 level. Attempts were made to add independent variables to WCI to improve ability to predict group membership. These attempts were rarely successful in making improvements and usually yielded statistically insignificant coefficients. An exception was adding riparian canopy cover rating for Puget Sound B-IBI ≤ 45 percent and ≤ 25 percent of best integrity, but the improvement was only marginal. This finding further verifies the utility of the WCI for defining watershed circumstances pertinent to stream biology.

Table 9. Logistic Regression Models Predicting Membership in Selected Biological Integrity Groups in Relation to Watershed Condition Index

LOCATION	BIOLOGICAL INTEGRITY GROUP [a]	b_0[a]	b_1[a]	b_1 SIG.[b]	b_{1CLl}[c]	b_{1CLu}[c]	CORR. NO (%)[d]	CORR. YES (%)[d]
Puget Sound	B-IBI ≥ 85% of best integrity	Neg. 10.6	0.103	<0.001	0.045	0.160	100	0
	B-IBI ≥ 75% of best integrity	Neg. 10.4	0.108	<0.001	0.059	0.156	97	27
	B-IBI ≤ 45% of best integrity	4.37	Neg. 0.078	<0.001	Neg. 0.102	Neg. 0.054	85	62
	B-IBI ≤ 25% of best integrity	1.34	Neg. 0.060	<0.001	Neg. 0.082	Neg. 0.037	96	29
	CS:CT ≥ 2.0	Neg. 8.45	0.132	<0.05	0.029	0.235	91	63
	CS:CT ≤ 1.0	5.77	Neg. 0.129	<0.01	Neg. 0.222	Neg. 0.035	93	86
Austin	Invertebrate score ≥ 85% of best integrity	Neg. 7.31	0.065	NS	Neg. 0.020	0.150	100	0
	Invertebrate score ≥ 75% of best integrity	Neg. 8.26	0.084	<0.05	0.002	0.166	100	20
	Invertebrate score ≤ 45% of best integrity	1.95	Neg. 0.033	<0.05	Neg. 0.060	Neg. 0.005	65	70
	Invertebrate score ≤ 25% of best integrity	1.42	Neg. 0.053	<0.01	Neg. 0.092	Neg. 0.015	87	44
Montgomery County	B-IBI ≥ 85% of best integrity	Neg. 6.45	0.061	<0.001	0.029	0.094	100 (100)[e]	0 (0)[e]
	B-IBI ≥ 75% of best integrity	Neg. 6.76	0.082	<0.001	0.053	0.111	83 (81)[e]	36 (47)[e]

Table 9 continued

LOCATION	BIOLOGICAL INTEGRITY GROUP [a]	b_0[a]	b_1[a]	b_1 SIG.[b]	b_{1CLl}[c]	b_{1CLu}[c]	CORR. NO (%)[d]	CORR. YES (%)[d]
Montgomery County	B-IBI ≤ 45% of best integrity	6.20	Neg. 0.095	<0.001	Neg. 0.121	Neg. 0.069	93 (89)[e]	67 (58)[e]
	B-IBI ≤ 25% of best integrity	3.63	Neg. 0.074	<0.001	Neg. 0.094	Neg. 0.054	94 (93)[e]	56 (40)[e]
	F-IBI ≥ 85% of best integrity	Neg. 3.54	0.031	<0.01	0.010	0.051	100 (100)[e]	0 (0)[e]
	F-IBI ≤ 60% of best integrity	1.91	Neg. 0.035	<0.001	Neg. 0.049	Neg. 0.020	91 (94)[e]	38 (30)[e]

[a] Probability, P, of membership in the given biological integrity group is computed as $P = e^L/(1 + e^L)$, where the logit $L = b_0 + b_1(WCI)$. Neg. signifies a negative number.
[b] Statistical significance of the regression coefficient b_1. NS signifies not significant at < 0.05.
[c] Upper and lower bounds of the probability estimate, P_u and P_l, are calculated as $P_u = CL_{Lu}/(1 + CL_{Lu})$ and $P_l = CL_{Ll}/(1 + CL_{Ll})$, respectively, where CL_{Lu} and CL_{Ll} are the upper and lower 95 percent confidence limits for the exponential of the logit, respectively. $CL_{Lu} = \text{Exponential}[b_0 + b_{1CLu}(WCI)]$; $CL_{Ll} = \text{Exponential}[b_0 + b_{1CLl}(WCI)]$.
[d] Percentage of values used to develop the model that would be correctly predicted as being a member of the group (corr. yes) or not being a member of the group (corr. no).
[e] The first number is the result from the randomly selected data used to develop the models (approximately half of the total data set). The number in parentheses is the result from the remainder of the data.

Prediction of <u>not</u> being in a group was more often correct than being a group member in all cases (with one minor exception). Hardest to forecast with these models would be very good benthic community health (≥ 85% of best integrity). In all three regions the models correctly placed all of the sites that did not fall in that group but none that did. Prediction of a very healthy fish community in Montgomery County exhibited the same result, although the Puget Sound model for at least a moderately intact coho salmon population was a better predictor. This consistent observation across regions is another reflection of the tendency seen and described earlier for a relatively high WCI to be a necessary but not sufficient condition for good biological health, with many points representing fairly natural watershed conditions still being degraded biologically. It may be seen in Table 9 that it was somewhat easier to predict membership in a moderately high integrity invertebrate group (≥ 75% of best integrity), although correct placements were still quite low.

In terms of placing streams in degraded biological groups, the models forecast correctly most points not falling into these groups. Their success was greater in general, although inconsistent, than the models for high integrity in predicting if a site would have the specified biological condition. The generally limited ability to predict group membership, in contrast to the greater success in forecasting exclusion from the group, makes this technique best suited to analyze if it is possible, with the existing or expected watershed condition, for a stream to achieve a high level of biological integrity or avoid a low level. The method is less reliable, and is not recommended, for assessing if the biological state actually will reach a certain level. The discriminant function analyses discussed above were more successful in judging actual membership in relatively high and low benthic community integrity groups. The techniques can be used in concert to assist in judging how likely a certain biological state is for a particular case.

The very large Montgomery County data set permitted randomly dividing it into subsets of approximately equal size and using one to develop models and the other to verify them. The last two columns of Table 9 show the comparative predictions for the original data and the verification subset (in parentheses). It can be seen that ability to forecast correctly was comparable for both subsets, offering verification of the models for the purpose recommended in the preceding paragraph.

Analyses presented in preceding sections showed that Austin and Montgomery County results usually exhibited weaker trends than the Puget Sound data. However, the logistic regression models were similar in effectiveness among regions. This finding indicates that, even with data scatter, applying a probabilistic technique to a regional database can be a useful resource management tool. Specifically, this device can assist in answering key questions about what is necessary in watershed protection to preserve healthy aquatic biology and prevent deterioration to the lowest levels.

Applications of the Logistic Regression Models

As an illustration of the use of the models, a range of land cover scenarios was posited for the Puget Sound region and converted to WCI values according to the scoring system in Appendix E. The logistic regression equations in Table 9 then computed the probabilities that stream ecosystems in watersheds with these WCIs would fall in the various biological integrity groups outlined in Table 9. High and low 95 percent confidence limits of the probability estimates were also calculated for the B-IBI groupings. The relatively small sample size for the coho salmon:cutthroat trout ratio resulted in wide confidence bands that were less useful for interpretation than in the B-IBI cases. Table 10 summarizes the outcome of the analysis. A probability > 0.50 is taken as the criterion for predicting that the biological state is in the group named. The inverse of the Table 10 probability (1 – probability) would then be the criterion for predicting that the biological condition would not be in the group.

Looking first at predictions of relatively good biological health, probabilities of B-IBI ≥ 75 percent of best integrity rise above 0.50 only if WCI is above 96 percent of the best

condition (and do not reach 0.50 for B-IBI ≥ 85 percent at all). However, as demonstrated in the development process, the logistic regression models are stronger in predicting that a site would not have a given biological state (represented by the inverse of the probability) than in forecasting that it would attain that condition. The inverse of the upper 95 percent confidence limit reaches the 0.50 level with WCI slightly less than 68 percent of best condition for B-IBI ≥ 85 percent, B-IBI ≥ 75 percent, and CS:CT ≥ 2.0. This result gives strong evidence of very low probability for relatively healthy invertebrate and fish communities with WCI much under 70 percent. The shaded regions in Table 10 associated with WCI in the range from 79 down to 57 percent indicate a region of rapid loss of prospects for high biological integrity.

Table 10. Predicted Biological Integrity Probabilities for Streams in Puget Sound Watersheds over a Range of Watershed Condition Indices

WCI[a]	B-IBI[a] ≥ 85%	B-IBI ≥ 85% HIGH 95% CL[a]	B-IBI ≥ 75%	B-IBI ≥ 75% HIGH 95% CL	B-IBI ≤ 45%	B-IBI ≤ 45% LOW 95% CL	B-IBI ≤ 25%	B-IBI ≤ 25% LOW 95% CL	CS:CT[a] ≥ 2.0	CS:CT ≤ 1.0
100	0.43	1.00	0.60	0.99	0.03	0	0.01	0	0.99	0
96	0.34	0.99	0.50	0.99	0.04	0	0.01	0	0.99	0
86	0.15	0.96	0.24	0.95	0.09	0.01	0.02	0	0.95	0.01
79	0.08	0.88	0.13	0.86	0.15	0.03	0.03	0.01	0.87	0.01
68	0.03	0.56	0.04	0.55	0.28	0.07	0.06	0.01	0.62	0.05
57	0.01	0.19	0.01	0.18	0.48	0.19	0.11	0.03	0.29	0.17
50	0	0.07	0.01	0.07	0.62	0.33	0.16	0.06	0.14	0.34
46	0	0.04	0	0.04	0.68	0.41	0.19	0.08	0.09	0.45
39	0	0.01	0	0.01	0.79	0.60	0.27	0.13	0.04	0.68
32	0	0	0	0	0.87	0.75	0.36	0.21	0.01	0.84
18	0	0	0	0	0.95	0.93	0.57	0.47	0	0.97
11	0	0	0	0	0.97	0.96	0.67	0.61	0	0.99
0	0	0	0	0	0.99	0.99	0.79	0.79	0	1.00

[a] WCI—watershed condition index; B-IBI—benthic index of biotic integrity; CL—confidence limit; CS:CT—coho salmon:cutthroat trout ratio

Turning to predictions of relatively poor biological health, the probability of B-IBI ≤ 45 percent of best integrity occurs around WCI = 55 percent of best condition. As demonstrated in the course of developing the logistic regression equations, this prediction is more reliable than a forecast of high integrity. This biological state becomes highly probable between WCI of 46 and 39 percent, the region where the lower 95 percent confidence limit reaches 0.50 (see the shaded area on Table 10). Decline of the coho salmon:cutthroat trout ratio to 1.0 is very likely in the same WCI vicinity (this prediction was accurate in 86 percent of the cases when applied to the

original data set; see shading). A very depleted benthic community (B-IBI ≤ 25 percent) becomes probable just under WCI of 20 percent (also shaded in Table 10).

This exercise illustrates the utility of regionally calibrated logistic regression equations to provide probabilistic estimates of the aquatic biological condition as a function of the composition of the contributing watershed. It bears emphasizing once again that any forecast of high biological integrity must be tempered by two important considerations. First, as pointed out several times, high WCI representing relatively land cover is not sufficient alone to assure good health. A second point, really a statistical manifestation of the first one, is that the lower 95 percent confidence limits for the probabilities of B-IBI ≥ 85 percent and B-IBI ≥ 75 percent (not shown in Table 10) are very low. On the other hand, forecast of actually falling to low levels of biological health with a given watershed condition has much more surety. A watershed manager having a goal of retaining a healthy aquatic ecosystem or avoiding a poor one must first assure the requisite land cover exists and then search for and preserve the other circumstances accounting for the existing state of health.

STRUCTURAL BMP INFLUENCE

Partial correlation analyses were performed to assess the effect of coverage by BMPs on the regional biological and habitat indices. There was no information on the quality of BMP implementation, only BMP areal coverage. WCI and TIA were separately assigned as control variables to remove their influence. Among the three regions, there was only one statistically significant, positive correlation between a biological or habitat metric and a measure of BMP coverage. The Montgomery County Fish IBI was correlated with BMP coverage at the $P < 0.05$ level of significance with WCI controlled, but the partial correlation coefficient was only 0.17. The significance disappeared, however, when the control variable was TIA. This analysis, therefore, shows no evidence that the mere presence of structural BMPs has any material influence on aquatic ecosystem quality.

Considering only BMP coverage, this analysis was very crude. A fair analysis of what structural BMPs can accomplish is possible only by incorporating quality as well as quantity of application, as was done earlier for a few of the Puget Sound watersheds and is described in the Background section of this report. However, collecting the necessary data for such an analysis is very labor intensive compared to GIS assessment of land cover and could never be performed for a broad selection of a region's streams. Nevertheless, additional targeted investigations like the earlier Puget Sound one may be fruitful. The Puget Sound evaluation suffered from not having any locations with high levels of coverage by BMPs designed and operated at present state-of-the-art standards. To show with some conclusiveness how much structural BMPs can protect urban stream ecosystems, future work of this type should focus on areas with are well served by high quality BMPs.

SUMMARY, CONCLUSIONS, AND RECOMMENDATIONS

RELATIONSHIPS AMONG WATERSHED CONDITIONS AND STREAM BIOLOGY

Defining Watershed and Biological Conditions

Coordinated studies in three different regions in the United States related stream biological communities to land use and land cover attributes of the watersheds draining to their habitats. Biological communities were defined in terms of multi-metric benthic macroinvertebrate indices developed for each region and indices characterizing fish communities in two regions. Initially, watershed land cover was defined in terms of total impervious area (TIA); the proportion covered by forest; and six variables describing the extent, continuity, and vegetative cover of riparian buffer zones. Later, geographic information system (GIS) analysis more specifically delineated watershed pervious and impervious cover. The intent with this classification was to represent not only the amount of these general land surface types but also their character and the activities occurring there.

GIS data were used to develop multi-metric Watershed Condition Indices (WCIs) for each region. Metrics comprising the WCIs are either relatively highly correlated to biological indices or were identified in preliminary stepwise multiple and logistic regression exercises as instrumental in linking watershed and aquatic biological states. Among all three regions the most prominent landscape variables relatively highly correlated to biological metrics are measures of total impervious area and forest cover at the watershed scale and in riparian buffer zones over a range of widths. The regression exercises pointed out, in addition, some instrumental features of local areas not necessarily in riparian zones but within certain distances near streams. WCI composition was fine-tuned by determining the combination of metrics giving the best linear or exponential model fits when biological indices were regressed on WCIs

General Observations

Graphical portrayals of biological indices versus measures of watershed attributes, both the initial set and the WCIs, were very useful in revealing a number of relationships between stream biota and the upland surroundings. Most striking is that the highest biological indices in all cases are associated only with the highest WCI values, representing no or extremely low urban development, very high forest retention, and minimal human intrusion in riparian zones. It was therefore demonstrated in three different regions of the nation that the best biological health is impossible unless human presence is very low <u>and</u> the natural vegetation and soil systems are well preserved near streams and throughout watersheds. However, while these conditions are necessary for high integrity, they are not sufficient by themselves to guarantee it. Other circumstances not captured in the GIS-based watershed analysis must also be instrumental.

An additional observation common among regions was that biological responses to urbanization in combination with loss of natural cover do not exhibit thresholds of watershed change that can be absorbed with little decline in health. Instead, decline was seen to start in the earliest stages of land conversion to human occupation. Rates of change in biology are relatively rapid in these early stages and then progressively slow with further urbanization. This pattern is probably a reflection of biological communities populated, more and more in the progression of human influence, with organisms reduced in variety but increasingly tolerant of additional stress.

Furthermore, in all three regions comparatively high urbanization and natural cover loss make relatively poor biological health the inevitable outcome. Thus, little or no urbanization and widespread preservation of natural land cover allows the existence of rich aquatic biological communities, although does not guarantee them. In contrast, extensive conversion to impervious and less pervious surfaces does guarantee depauperate ecosystems.

Along with these common general trends among regions, there is a fair degree of unity in the specific watershed conditions associated with the highest and lowest levels of biological integrity. Taking at least 80 percent of best integrity as an example definition point for good benthic invertebrate community health, WCI in the range of 70 to 80 percent of best watershed condition is essential in all three regions to attain this biological state. A watershed index at least at the lower end of this range is also necessary for clear dominance (in the ratio of at least 3:1) of the over-wintering Puget Sound salmonid fish community by coho salmon instead of the more tolerant cutthroat trout.

At the opposite end of the biological spectrum, poor invertebrate community health (taken for example as under 40 percent of best integrity) occurs in each region, excepting only two cases, at WCI = 25-30 percent of best condition and below. Cutthroat trout dominance is also assured under these watershed conditions in the Puget Sound streams.

Quantification of Results

Several statistical and multivariate analytical techniques were applied to evaluate the Watershed Condition Index and devise formal mathematical constructs to increase its utility as an assessment and management tool. Discriminant function analyses validated the regional WCIs, and, independently, their component variables, as mechanisms for classifying biological integrity according to watershed condition. Sites in the highest and lowest integrity groups were generally more successfully classified in these analyses than those in the intermediate categories.

A second multivariate technique applied to the data was logistic regression analysis. This analysis produced equations forecasting the probability of a stream's invertebrate or fish community being in selected groupings of biological integrity based on WCI:

$$P = e^L/(1 + e^L) \qquad \text{and} \qquad L = b_0 + b_1(WCI)$$

where: P = Probability of membership in a given biological integrity group (> 0.5 to assign membership);
e = Base of natural logarithms;
L = Logit function;
b_0 and b_1 = Constant and logistic regression coefficient derived in the analysis, respectively (Table 9); and
WCI = Watershed condition index (% of best condition).

When applied to the original data and an independent Montgomery County data set held aside for model verification, the equations were more successful in predicting that a site would not have a certain biological condition than forecasting that it would fall in the specified group. Hardest to forecast with these models is very good benthic community health. This consistent observation across regions is another reflection of the necessity but not sufficiency of relatively high WCI for high biological integrity, with many points representing fairly natural watershed conditions still being degraded biologically. The models were more successful, although still inconsistent, in predicting membership in degraded biological groups than in high quality categories.

The generally limited ability of the equations to predict group membership, in contrast to the greater success in forecasting exclusion from the group, makes this technique best suited to analyze if it is possible, with the existing or expected watershed condition, for a stream to achieve a high level of biological integrity or avoid a low level. The method is less reliable, and is not recommended, for assessing if the biological state actually will reach a certain level. The discriminant function analyses discussed above were more successful in judging actual membership in relatively high and low benthic community integrity groups. The techniques can be used in concert to assist in judging how likely a certain biological state is for a particular case. It must always be recalled, though, that actual achievement of the best biological health depends on some factors yet to be defined.

To bring in a more formally quantitative view supplementary to the earlier graphical observations, the Puget Sound logistic regression equations for the macroinvertebrate (B-IBI) and fish communities (coho salmon:cutthroat trout ratio) were applied to hypothetical watershed conditions. The results give strong evidence of very low probability for relatively healthy invertebrate and fish communities with WCI much under 70 percent, a conclusion agreeing with the graphical interpretation. WCI in the range from 79 down to 57 percent of best condition is a region of rapid loss of prospects for high biological integrity. B-IBI ≤ 45 percent of best integrity is highly probable as WCI goes below 45 percent. Decline of the coho salmon:cutthroat trout ratio to 1.0 is very likely around the same WCI. A heavily depleted benthic community (B-IBI ≤ 25 percent) becomes probable just under WCI of 20 percent. These tendencies too echo those observed on the graphs.

The more clear-cut results at relatively high compared to low urbanization render these methods most useful in the more urban areas to analyze how to prevent already deteriorated biological integrity to even lower levels, or to improve health somewhat. They can also be applied at very low urbanization, but only with the clear realization that favorable watershed conditions are necessary but not sufficient for confidently predicting good biological health.

THE ROLE OF STRUCTURAL STORMWATER BEST MANAGEMENT PRACTICES IN STREAM BIOLOGICAL INTEGRITY

Extensive and incisive investigation of how stormwater BMPs affect the portrait of aquatic biology in relation to overall watershed conditions was hindered by the very labor-intensive effort required to collect meaningful data on the numerous BMPs that often exist in urban watersheds. For example, the first approximately 40 stream basins or subbasins studied in the Puget Sound region have over 2600 BMPs. Meaningful evaluation would require detailed data of various kinds on BMP siting, design, and operation in relation to stormwater management objectives and contributing catchments. With this dilemma the study proceeded in two directions: (1) a broad approach over all watersheds with recorded BMP presence to determine if the mere extent of BMP coverage, with no assessment of implementation quality, has an identifiable, positive effect on stream health; and (2) a deeper effort in a few watersheds to collect and evaluate the data necessary to gauge BMP implementation quality and its effect on aquatic systems.

The broad-scale approach was not very fruitful. Early graphical plots of biological versus urbanization measures for catchments with and without BMPs did not distinguish differences in biological quality between the two groups. Follow-up statistical examinations of BMP areal coverage expressed in several ways (e.g., per km^2, per unit of impervious cover), with overall watershed condition being a controlled variable, exhibited very weak or even negative partial correlations between biological integrity and BMP presence.

In the second, deeper approach, structural BMPs were intensively studied in several subbasins of two Puget Sound stream systems, one with perhaps the greatest consideration to stormwater management in the region and the other with less attention. Even in the first watershed, a minority of the developed area is served by runoff quantity control practices, and even less of it by water quality control BMPS. Much development was vested with approvals before BMP requirements took effect or was exempted on the grounds of falling below development size thresholds. Those BMPs installed are capable of mitigating an even smaller share of urban impacts, primarily because of inadequacies in design standards.

Even with these shortcomings, though, results indicate that structural BMPs appear to help in sustaining aquatic biological communities at fairly high urbanization levels. They give less evidence of benefit at moderate urbanization and greater natural land cover. If ecological losses are to be stemmed at high urbanization, structural BMPs appear to

have a substantial role. In this situation development has taken forests and wetlands and intruded into riparian zones, reducing their roles in the watershed. In the most urban areas it seems that these roles can be assumed in part, but not in full, by structural BMPs.

The highest biological indices had no relationship to BMPs, because these high scores occurred only in watersheds with no or minimal development, where no BMPs were built. It thus could not be tested if BMPs can replace some loss in natural land cover through light urbanization and still maintain high biological integrity. However, the lack of obvious benefit seen with a moderate amount of development lends support to the hypothesis that the benefit would also be absent at low urbanization too, where relatively undisturbed streams house the most sensitive organisms.

Any conclusions from this analysis must be tempered according to the scope of the underlying data. There were no instances found of structural BMPs being exceptionally widely applied and designed to mitigate a large share of the known impacts of urbanization. Therefore, the fullest potential of these practices has not been examined, and it is possible that extremely thorough applications would demonstrate additional benefits not suggested in these data.

RECOMMENDATIONS

Unity in the results from three dispersed and differing areas of the nation support certain general watershed management recommendations for strong consideration elsewhere. This work also developed methods that can be broadly recommended to assist any region wishing to develop a basis for its own watershed analysis and management efforts.

General Watershed Management Recommendations

1. Base watershed management on specific objectives tied to desired biological outcomes.

2. If the objective is to attain an existing high level of biological integrity, very broadly preserve the extensive watershed and riparian natural vegetation and soil cover almost certainly present through mechanisms like outright purchase, conservation easements, transfer of development rights, etc.

3. If the objective is to prevent further degradation when partially developed areas urbanize further, maximize protection of existing natural vegetation and soil cover in areas closest to the stream, especially in the nearest riparian band. In the uplands, generally develop in locations already missing characteristic natural vegetation. As much as possible, preserve existing natural cover and limit conversion to impervious surfaces. The lower the level of existing development, the more important this recommendation is.

4. In addition, fully serve newly developing and redeveloping areas with stormwater quantity and quality control BMPs sited, designed, and operated at state-of-the-art levels. Attempt to retrofit these BMPs in existing developments. The higher the level of existing development, the more important this recommendation is; since much opportunity to apply the preceding recommendation is lost with extensive land conversion.

5. Where riparian areas have been degraded by encroachment, crossings, or loss of mature, natural vegetation, give high priority to restoring them to extensive, unbroken, well vegetated zones. This strategy could be the most effective, as well as the easiest, step toward improving degraded stream habitat and biology. Riparian areas are more likely to be free of structures than upland areas and more directly influence stream ecology. Also, riparian restoration fits well with other objectives, like flood protection and provision of wildlife corridors and open space

6. The above recommendations suggest that federal and state environmental management agencies should reconsider their existing water body classification systems and the associated water quality standards. This is consistent with the recommendation of the National Academy of Sciences (NRC 2001) review of the nation's total maximum daily load (TMDL) program that states needed to conduct use attainability analyses and appropriate designate the beneficial uses of water bodies. State watershed managers need to work closely with local communities to develop water body classifications that accurately reflect the desired and achievable goals of the community for its aquatic ecological systems.

Recommendations for Developing Regional Watershed Analysis and Management Approaches

1. Systematically collect data on regionally representative stream benthic macroinvertebrate and fish communities. Extend the program's coverage over the full range of urbanization, from none to the highest levels with above-ground streams. Use the data to develop regionally appropriate biological community indices.

2. Develop a geographic information system to organize and analyze watershed land use and land cover (LULC) data. Collect data on regionally appropriate LULC variables, particularly measures of impervious and forested cover in the watershed as a whole, at least two riparian bands extending to points relatively near and far from the stream, and in other local areas fairly close to the stream.

3. Investigate which LULC variables are statistically best associated with biological indices, using analyses like correlation and stepwise multiple and logistic regressions.

4. Define a tentative Watershed Condition Index (WCI) using the best associated variables.

5. Choose the optimum LULC variables for the WCI on the basis of the combination yielding the best fits in statistical regressions of biological indices on WCI. These regressions are useful for fine-tuning the WCI but are unlikely to offer very good tools for predicting biology as a function of WCI.

6. Validate the resulting WCI with discriminant function analyses as described in this report.

7. Graph biological indices versus WCI and examine trends signifying potentially fruitful regional watershed management strategies.

8. Perform logistic regression analyses to develop means of classifying probable groupings of aquatic biological health in relation to WCI. This type of analysis was found in this study to be better at predicting if a particular case would not be in a group than if it would be.

9. Supplement the logistic regressions with discriminant function analysis, which was found to be better at forecasting if a case would fall in a group.

10. Use the two techniques in concert to make judgments like: (1) With prevailing or expected watershed conditions, is it possible for a biological state to be at the highest level or, in other situations, avoid the lowest level? (2) With these conditions, how likely is it that the state will actually attain that level? (3) What management strategies can be considered, and are most likely to be feasible and successful, to adjust watershed conditions in a way that will maximize the chance of attaining a biological objective?

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APPENDIX A

BENTHIC INVERTEBRATE SAMPLING AND ANALYSIS PROCEDURES

PUGET SOUND PROCEDURES

Sampling Protocol

Checklist of materials:
__ Meter tape to identify location
__ 500 micron mesh Surber sampler
__ (2) 500 micron mesh sieve (or smaller)
__ Waders (for each person)
__ Flagged weight to identify sample location
__ Isopropyl alcohol
__ 1-liter squirt bottle for isopropyl alcohol; second bottle to refill the first
__ Garden trowel to disturb substrate
__ Stop watch
__ (2) White buckets to empty sample from Surber
__ Large cup with handle to rinse invertebrates off Surber
__ Forceps (Tweezers)
__ Plastic spatula
__ Waterproof ("Rite-in-the-rain") paper
__ Pencil, permanent marker (Sharpie), and grease pencil
__ Screw-top vials
__ Ziploc bags
__ Camera

Select site:

Locate stream reach to be sampled. Find a riffle (fast moving water over rock or cobble substrate, surface water should be broken) near the middle part of the stream. Riffle should be long enough to accommodate three replicate samples. Ideal sampling locations consist of rocks 5 to 10 cm in diameter sitting on top of pebbles. Substrates dominated by rocks larger than 50 cm in diameter should be avoided.

Sample within main flow of the stream. Sample at water depths of 10 to 40 cm. Depth, flow and substrate type should be similar for the three replicate samples collected in the riffle. Begin sampling downstream and proceed upstream for the three replicates. Avoid bridges and other large human-made structural features. If unavoidable, sample at least 50 meters upstream of a bridge and 200 meters (more would be better) downstream of a bridge.

Write down the exact location of the sample site. Use meter tape to measure distance from nearest landmark.

Collect invertebrates:

Sampling teams may range from 2 to 4 people. Actual collection of macroinvertebrates requires 2 people. Others can assist with equipment, labeling collections and other duties.

1. Place Surber sampler on the selected spot with the opening of the nylon net facing upstream. Brace the frame and hold it firmly on the creek bottom.
2. Lift the larger rocks resting within the frame and brush off crawling or attached loosely organisms so that they drift into the net. After 'cleaning' the rocks, place them in a bucket.
3. Once the larger rocks are removed, disturb the substrate vigorously with a trowel or large spike for 60 seconds. This disturbance should extend to a depth of about 10 cm to loosen organisms in the interstitial spaces, washing them into the net.
4. Lift Surber out of the water: Tilt the net up and out of the water while keeping the open end upstream. This helps to wash the organisms into the receptacle. Drop a piece of weighted flagging tape to mark the location of the first replicate sample.
5. On the creek bank, empty contents of Surber into large bucket. Rinse Surber and empty into bucket until all animals are removed. Great care should be taken in this step to collect and preserve all organisms from the Surber sampler as well as from the rocks and water in the bucket. Use of a magnifying glass and tweezers is essential. Rinse bucket through sieve to remove water from sample. Pick out large debris (sticks and leaves) after carefully removing any invertebrates.

Archive sample:

Use spatula to move sample from sieve into a plastic vial. Fill vial to the top with isopropyl alcohol. Put label on inside of vial with name of sampler, date, location, and replicate number. Write location and date on top of vial lid. Place vial in a Ziploc bag labeled with the same information.

Collect replicate samples:

Return to the location of the first sample, walk upstream and collect another sample of invertebrates. Leave another flagged marker and process the sample as above. Repeat this process once more for a total of three replicate samples from each site location. Each replicate should be labeled (e.g., #1, #2, #3) and archived separately.

Taxonomic Identification and Benthic Invertebrate Index (B-IBI) Scoring

Note: This specific procedure was used for samples collected after 1998. A slightly modified procedure was used for earlier samples (1994-1998). This procedure used nine instead of ten metrics, eight the same as the list below (excluding number of clinger taxa and percent dominance) plus percent planaridae and amphipoda

abundance. The two different systems were reconciled for data analysis, as well as reconciled with similar indices from Austin and Montgomery County, by placing all indices on the basis of "percent of best integrity" as described in the report section Reconciling Indices Among Regions for Comparison Purposes under Methods for Comprehensive Data Analysis.

Taxonomy and Metric Development:

Identify all aquatic insects to the species level, except chironomids, which are identified to the genus level, and rhyacophilids, which are identified to subgroup. Non-insects are identified to the order or family level. Define the following metrics:

Total Taxa Richness

The total number of unique taxa identified in each replicate. The numbers from the three replicates are then averaged for this metric.

Ephemeroptera Taxa Richness

The total number of unique mayfly (Ephemeroptera) taxa identified in each replicate. The numbers from the three replicates are then averaged for this metric.

Plecoptera Taxa Richness

The total number of unique stonefly (Plecoptera) taxa identified in each replicate. The numbers from the three replicates are then averaged for this metric.

Trichoptera Taxa Richness

The total number of unique caddisfly (Tricoptera) taxa identified in each replicate. The numbers from the three replicates are then averaged for this metric.

Number of Long-Lived Taxa

The total number of unique long-lived taxa identified in each replicate. The numbers from the three replicates are then averaged for this metric.

Number of Intolerant Taxa

The total number of unique intolerant taxa identified in each replicate. Chironomids are not included in this metric. The numbers from the three replicates are then averaged for this metric.

Percent Tolerant Individuals

The total number of tolerant individuals counted in each replicate, divided by the total number of individuals in that replicate, multiplied by 100. Chironomids are not included in this metric. The numbers from the three replicates are then averaged for this metric.

Number of Clinger Taxa

The total number of unique clinger taxa identified in each replicate. The numbers from the three replicates are then averaged for this metric.

Percent Predator Individuals

The total number of predator individuals counted in each replicate, divided by the total number of individuals in that replicate, multiplied by 100. The numbers from the three replicates are then averaged for this metric.

Percent Dominance

The sum of individuals in the three (3) most abundant taxa in each replicate, divided by the total number of individuals in that replicate, multiplied by 100. The numbers from the three replicates are then averaged for this metric.

Percent Dominance Example

Step 1 **Calculate taxa totals**	**Step 2** **Sum 3 Most numerous Taxa**	**Step 3** **Calculate Percentage**
Taxon 1 = 10 organisms Taxon 2 = 8 organisms Taxon 3 = 3 organisms Taxon 4 = 1 organism	Pick Top 3: Taxa 1 = 10 Taxa 2 = 8 Taxa 3 = 3	(# organisms in 3 dominant taxa / Total # individuals) X 100 (21 / 22) X 100
Total = 22 organisms	Total = 21 organisms	Percent Dominance = 95%

Scoring Criteria:

Criteria are for species-level identification of most insects, rhyacophilids to subgroup, and chironomids to genus. Square braces indicate the value next to the brace is included in the range; rounded parentheses indicate the value is not included.

Metric	Scoring Criteria 1	3	5
Taxa richness and composition			
Total number of taxa	[0, 20)	[20, 40]	> 40
Number of Ephemeroptera (mayfly) taxa	[0, 4]	(4, 8]	> 8
Number of Plecoptera (stonefly) taxa	[0, 3]	(3,7]	> 7
Number of Trichoptera (caddisfly) taxa	[0, 5)	[5, 10)	≥ 10
Number of long-lived taxa	[0, 2]	(2, 4]	> 4
Tolerance			
Number of intolerant taxa*	[0, 2]	(2, 3]	> 3
% of individuals in tolerant taxa*	≥ 50	(19, 50)	[0, 19]
Feeding ecology			
% of predator individuals	[0, 10)	[10, 20)	≥ 20
Number of clinger taxa	[0, 10]	(10, 20]	> 20
Population attributes			
% dominance (top 3 taxa)	≥ 75	[50, 75)	[0, 50)

*****Chironomids are not included in these metrics.

Worksheet:

METRICS (1, 3, or 5)	**Rep 1**	**Rep 2**	**Rep 3**	**Replicate Average**	**Metric IBI Score**
Total number of taxa					
Number of Ephemeroptera (mayfly) taxa					
Number of Plecoptera (stonefly) taxa					
Number of Tricoptera (caddisfly) taxa					
Number of long-lived taxa					
Number of intolerant taxa					
% of individuals in tolerant taxa					
Number of clinger taxa					
% of predator individuals					
% dominance (3 taxa)					
Total B-IBI Score (Add Metric B-IBI scores for Total B-IBI score)					

For the percentage metrics, remember to multiply the final computation by 100 for each replicate, e.g., % predator individuals = (total number predator individuals / total number individuals) X 100.

AUSTIN PROCEDURES

Standard Operating Procedures for Biological Sampling Protocols

(For all surveys: Count, note and remove all *Corbicula* from samples and put permanent paper labels in all field vials.)

1. Screening level/EII Surveys

- All field teams will bring with them the following: one Surber sampler (1 ft^2, 600 µm mesh-size), one Caton subsampler, sample vials and a picking pan for each person who will be sorting.
- Three Surber samples, including all detritus, will be collected in the bag from the bottom, middle and top parts of the riffle and composited in the subsampler, distributing material evenly throughout grid.
- At this point, abundance is noted:
- ➢ For high abundance samples (>1000 organisms), one grid (out of 30) will be randomly selected which will be removed and transferred to the picking pan to be picked in it's entirety. Sequential grids will be picked until the target number of 200 organisms is reached (+or- 20%).
- ➢ Lower abundances will require more than one grid per person picking (2-9). As a guide, if each grid has < 7 organisms, you will have to pick the whole pan to get your target. Think rapid and pull enough grids to give each picker ~25 – 50 orgs. These adjustments can be made as you go.
- ➢ Extremely low abundance samples can be picked in their entirety and supplemented with sequential Surbers until the target number of organisms is reached (200 +/- 20%). However, be sure to maintain the original 3 Surber composite discrete, and each subsequent Surber after that discrete as well (3-Surber composite = 1 sample in DB; each subsequent Surber = 1 sample in DB. Always record # of Surbers, # of grids, Total # of estimated orgs)
- The number of grids/Surbers subsampled is noted along with the estimated number of organism in each grid/Surber. We need to be able to document level of effort (area) to reach our target number (200).
- Have each sorted pan (grids or Surbers) reviewed by a different field staff than the one who picked it for quality control.
- Preserve the detritus from 1 out of every 10 samples and have them lab verified to monitor our percent recovery.

2. Intensive Surveys

- Pick three discrete Surbers entirely from the bottom, middle and top of the sample riffle. No fixed count. No sub-sampling. Label samples so they can be spatially located (a, b, c…).
- Have each sorted pan (grids or Surbers) reviewed by a different field staff than the one who picked it for quality control.
- Preserve the detritus from 1 out of every 10 Surbers and have them lab verified to monitor our percent recovery.

- In case of an excessively productive site (1000+ orgs per Surber), each Surber is treated as above, subsampling +4 grids per Surber or 100 orgs, which ever comes first (for a total of 300+ orgs). Upon completion of the subsampling each Surber is reviewed by the senior taxonomist for large and rare organisms, which are stored in a fourth vial for calculating richness/diversity metrics. Again, note number of grids sorted and number of orgs per grid for level of effort extrapolation. Maintain three discrete replicates plus large/rare sample for analysis purposes (= 4 samples in DB)
- In case of an excessively depauperate site, after the first three Surbers have been sorted (3 samples in DB), successive Surbers (one sample each) may be collected and sorted if the field staff desires to increase total number of organisms to 100 (+/- 20%) in order to calculate. However, this is not required and the three-Surber sample with uniform area corresponding to other sites should always be the base data source for analysis purposes.

Note: If Intensive and EII surveys have crossover and the data will need to be used for both purposes, the Intensive survey method should be used. If the site has less than 200 organisms, successive Surbers should be collected (discretely, one Surber = one sample in DB) until at least 200 organisms is collected. If this is not practical (extremely low abundance), retain each Surber that has been collected (+3 discrete samples in DB) and note that site did not reach target number in field notes. When the total number of orgs is higher than 200, it will be mathematically reduced to the target number for EII analysis.

3. Reference Sites:

Background: We are compiling historical data and establishing a seasonal and regional reference condition using multiple sites/dates. It will provide us with the character and type of variability that we would expect at reference sites, which can then be used to normalize metrics and other types of analysis. There were no significant differences between East and West reference site metrics, so we will use a composite of both regions for our analysis. We will segregate seasons loosely and see if there is difference in metric values in Spring/Summer vs. Fall/Winter conditions. We will continue to collect reference data according to the following protocol in order to check our surveys for outliers, continue building our database and provide values for the metrics that require a specific reference comparison.

- The following bank of reference sites has been selected based on staff experience and percent impervious cover. Each of these sites should be sampled during the two sampling seasons when intensive surveys are carried out (Spring and Summer) and during the EII surveys.

    1. Barton Creek at Shield Ranch (West)
    2. Barton Creek at Hwy 71 (West)
    3. Bull Creek at Franklin's Tract (West)
    4. Bull Creek at St. Ed's Park (West)
    5. Onion Creek at Hwy 150 (West)

6. Onion Creek at Twin Creeks (East)
7. Onion Creek at South Austin Regional (East)
8. Walnut Creek at Old Manor (East)

Note: We don't really have good Eastern Reference sites. None of our sites are true "Blackland Prairie". They are more transitional, so we can't really use state reference data. In addition, all our sites East of I35 are pretty degraded by urbanization or agriculture. This is a problem.)

4. Reiteration of sampling strategy suggestions:

a) We should seriously consider structuring our biological sampling to fit the analysis goals of our studies. What this means is that we don't do quarterly sampling just because it seems like a good idea and because we always did it that way. If we want to look closely at spatial or temporal changes our analysis should address that goal. We should do multiple sampling during one season for spatial variation. We should do intensive, quantitative sampling (lab sorted) 2-4 times per year for a max of 3 years if we want to do temporal analysis or document year round community structure (note: this is an academic question, not a water quality question).
b) We should do a qualitative sampling regime (see method 1) for EII and probably would be better off with a kick net that will sample a larger area (than 3 sq. ft). This should be composited and picked in the field for speed to a target number of orgs. 100 would be defensible if we are using this data for routine indexing. 200 would be better, but not a lot better…(according to several recent literature sources).
c) I think we should reduce our sampling frequency on our "intensive" creeks to 2 times per year. 2x in the late spring/summer (indexing period) or in the winter and summer if we want to try and document seasonal communities (why?).
d) Our "intensive" sampling should have more specific monitoring goals. If we really want to nail down community structure or pick apart small differences in sites, we should be doing true quantitative sampling. We would do three discrete or composited Surbers (Jack Davis likes composited) and sort all orgs in the lab and use metrics which are designed for quantitative data. If we just want to keep track of long term temporal trends, or just keep our eye out for problems, we can get by with annual, more qualitative surveys (EII method).
e) With the current intensive survey protocol (three field picked Surbers), we can start evaluating our within site variability and see if we want to go with more area or a different method (kick-nets, lab sorting…). If our within site variability is greater than our between site variability, than we can't say anything about differences between sites. If we sample more area (kick nets) we may be able to get a more representative site sample that holds up better in statistical analysis

Index Development Using a Percentile Reference Condition

1. Metric scores from all available EII data are used to characterize the natural background distribution in any given metric score. From this data set, the 95th and 5th percentiles are used to define the range of values within the data set, excluding

outliers (Table 1). For example, the HBI raw metric score ranges from 1-10, but in our area is generally between 4 and 7. The 95th percentile of all the EII sites is 7.00. The 5th percentile is 4.68.

Table 1. Calculated Percentiles for 9 Metrics from All Available EII Benthic Macroinvertebrate Data

Proposed EII Metrics	DB#	# of obs	95th %	5th %
HILSENHOFF BIOTIC INDEX		270	7	4.68
NUMBER OF EPHEMEROPTERA TAXA		270	5	0
NUMBER OF EPT TAXA		270	8	0
NUMBER OF INTOLERANT TAXA		270	11	0
NUMBER OF TAXON		270	30	4
PERCENT DOMINANCE (TOP 3 TAXA)		264	98.07	46.57
PERCENT OF TOTAL AS CHIRONOMIDAE		270	87.5	1.09
PERCENT OF TOTAL AS EPT		270	65.92	0
PERCENT OF TOTAL AS PREDATOR		270	84.38	0

2. The 95 and 5th percentiles for each metric are then used to define the expected range which test /degraded sites are evaluated within and to truncate outlier scores. If a metric score is outside of 5th –95th range it is set to the closest one. The truncated value for each raw metric is used to index EII scores using the following formula:

   EII Metric Score = 100 * (Truncated raw metric score – 5th percentile/95th percentile – 5th percentile).

   This procedure is repeated for each metric and then the average EII metric score is calculated to get a site score. NOTE: For some metrics (% Predator, % Dominant, % Chironomidae, HBI) the 95th percentile is actually the worst score and the 5th is the best score. These metrics will be inverted to reflect a more intuitive 0-100% range by subtracting the final score from 100.
3. After getting an average score for each site, the bank of reference sites that are sampled during the same survey as the data in question is used to standardize that data to temporal constraints like season or flow. Among the reference sites from

any given survey, the site with the highest average score (all 9 EII metric scores averaged together) is used as the highest attainable score for that survey. All "test" sites are then compared to that. For example, Onion Creek at Pfulman Ranch got a total site score of 89%, which is the highest of all reference sites sampled. If Lake Creek at Sugarberry got a 72 during that survey, it would be adjusted using 89 as the max possible, resulting in: 72/89= 81. Onion at Pfulman would be raised to perfect (100) because it got the highest score and all other sites would be adjusted relative to that score. Any site getting better than the maximum reference site would achieve a score of 100.

4. I am including a spreadsheet which demonstrates how this procedure would be applied using the 9 proposed EII metrics and one test/impaired site (Table 2).

Table 2. Example Calculations for Proposed Indexing System

	Metric	95th %	5th %	Test Site	Raw EII score	Ref. Site	Ref. % score	Indexed EII Score
1	HBI	7	4.68	5.71	0.56	5.05	0.84	
2	# of Ephem.	5	0	2	0.40	7	1.00	
3	# of EPT	8	0	6	0.75	7	0.88	
4	# of Intol. Taxa	11	0	6	0.55	6	0.55	
5	# of Taxa	30	0	15	0.50	28	0.93	
6	% Dom/3 taxa	0.98	0.47	0.67	0.61	0.51	0.92	
7	% Chiron.	0.88	0.01	0.33	0.63	0.05	0.95	
8	% EPT	0.66	0	0.36	0.55	0.6	0.91	
9	% Predator	0.84	0	0.51	0.39	0.1	0.88	▼
		Totals/Average			**54.8**		**87.3**	**62.7**

So, the raw site score is 54.8, but after adjusting it using the reference site from that survey (87.3) it gets a score of 62.7. I guess this would make it a Good site (Barely, the cutoff is 62.5…).

MONTGOMERY COUNTY PROCEDURES

Field Methods

All benthic macroinvertebrate inventory stations will be located within or immediately upstream or downstream of the sampling segments selected for fish and habitat inventory. Macroinvertebrate stations will include one riffle-pool-riffle reach at a minimum. Stations will be sampled two times a year during the spring (March 15 to April 15), and fall (October 15 to November 15) seasons.

The biologically optimal sampling periods for this region include late March through mid-April, July through August, and during late October through the first 2 weeks of November. The biologically optimal sampling period is the time when most of the population is within a size range that can be retained in a sieve and identified with the most confidence. These seasonal periods also correspond to periods when point and nonpoint source effects are most evident.

Within each sampling segment, benthic macro invertebrates are collected using a 1 m^2 kick net with a mesh size of 530 microns. Kick net inventory methods follow Plafkin et al. (1989), Primrose (1993), and Stribling (1994). Riffles are sampled using the kick net to collect from an approximately 2 m^2 composite area. Two 1 m^2 kick net samples will be collected per station; one from an area of fast current velocity and one from an area of slower current velocity (Plafkin et al. 1989: Stribling 1994).

Place the net so that it is fully extended. Disturb the substrate immediately upstream of the net for a 1 m^2 area. Large stones should be hand rubbed to dislodge attached macroinvertebrates (Primrose 1993). The substrate should be disturbed down to a depth of several inches, generally to the underlying stable or compacted strata of sand, rock, or clay (Primrose, 1993). No time limit is placed on this procedure, but generally it should take no more than 5 minutes. Care should be taken during sampling to maintain water flowing through the net, and to avoid a backflow situation. Backflow of water could carry organisms out of the collecting net, possibly biasing the sample (Primrose 1993).

The kick net is gently raised up to allow any remaining water to drain through the net to minimize sample loss (Primrose 1993). Each sample is washed down the kick net into either a #30 sieve bucket or nested sieves, with a #30 mesh sieve on the bottom. The use of nested sieves allows quick separation of the large stones and organic material from the sample. A sieve bucket can also be used. All large material is examined for organisms before discarding.

The second kick net sample is composited with the first (Plafkin et al. 1989, Stribling 1994). Large sticks, leaves, and fine sediments are picked and washed out of the composite sample. The entire composite sample is placed in a jar and emersed in 70% ethanol for later processing at the laboratory. The jar will be labeled with the date, name of the watershed, stream segment number, collector's name, air temperature, and water temperature.

Equipment and Materials:

1 m^2 kick seine net (#30 mesh)

Nested sieves (bottom sieve will be a #30 mesh sieve)

Ethanol (70% solution)

500 ml collection jars

4 to 6 dram screw cap glass vials

10.5x to 45x stereomicroscope

Identification manuals

Tweezers

Laboratory Methods

The composite sample is emptied into a gridded subsampling tray containing water and the sample is evenly distributed. Grids number 20 squares per pan. The smaller the square, the more random the subsample picking will be (Primrose 1993). As grids are randomly selected, all macroinvertebrates within the selected grids are removed. The grid that contains the 200th organism will be completely picked, and all the organisms in this last grid placed in the subsample for identification (Stribling 1994; Primrose 1993).

All macroinvertebrates in the subsample will be identified to genus, or to the lowest positively identified taxonomic level (Plafkin et al. 1989). Identification is made using a 10.5x to 45x zoom stereomicroscope. Fresh alcohol may have to be added to replace discolored alcohol.

The primary taxonomic key for taxonomic identification to the genus level is Freshwater Macroinvertebrates of Northeastern North America (Peckarsky et al. 1990). Other keys will be used as needed.

All identified samples will be maintained in a voucher collection for a period not less then two complete sampling periods (usually three to six years). A cumulative reference collection and list of benthic macroinvertebrates will be maintained by DEP. Additions to the cumulative list are invited, and will be accepted with the submission of a properly identified reference specimen. Required submittal information should include name of researcher, date of collection, location of collection, and identification of the specimen.

Data Management

The identified macroinvertebrate collection is organized taxonomically into a summary table. Identified taxa are listed in column format. Also to be included are columns for relative frequency of each taxon, trophic level of each identified taxon, and the tolerance to stream degradation of each identified taxon. The header description for each sample will include information for station location, date of sample, date of identification, collector's name, identifier's name, total number of organisms per composite sample, air temperature, and water temperature. Optional information will include time to the last storm event, and habitat observations.

The Department will design and implement a standardized, geographically relational database. The Department is currently supporting the use of Statistica 5.1 software for normal or routine management of the database.

Development of Biological Integrity Classes Based on a Reference Condition

Reference stream station IBI scores were graphed according to stream order and soil type. The median, 25th percentile, and 75th percentile values were calculated for each IBI grouping. IBI scores from the median score of the reference stations to the maximum score were assigned the highest attainable biological integrity class (of excellent). This range is very small (usually 4 points) and will be used as a conservative narrative classification. The range of IBI scores below the median would be indicative of declining biological integrity and declining water quality.

The benthic macroinvertebrate IBI median scores for 1-2 and 3-4 order reference stream datasets, as stratified by soil types, were within 3 to 4 points of each other. Because median scores were so close, the lowest median score was selected and the scores below this median were trisected. Narrative classes depicting the decline *of* biological integrity and water quality as IBI scores descended from the median score were added. The IBI score at each trisection breakpoint is assigned the next narrative level of biological integrity. The first range *of* scores below the median is assigned a narrative value of good. The next range is assigned a narrative value of fair. The last range of scores is assigned a narrative value of poor. Refer to the tables below for the scoring criteria for the various soil regimes and stream order classes.

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Primrose, N. 1993. Description of the Marvland Department *of* the Environment's Field Methods for the Inventory of Benthic Macroinvertebrates. Personal correspondence.

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Scoring Criteria for Channery Silt Loam Region, 1st and 2nd Order Streams

Scoring Criteria for Macroinvertebrate IBI Metrics			
	SCORING CRITERIA[a]		
METRIC	5	3	1
TAXA RICHNESS	>22	11 - 21	<11
BIOTIC INDEX	<3.31	3.31 - 6.66	>6.66
RATIO OF SCRAPERS/ (SCRAPERS+FIL TERING COLLECTORS)	>42%	21% - 42%	<21%
PROPORTION OF HYDROPSYCHE AND CHEUMATOPSYCHE/ TOTAL EPT INDIVIDUALS	<9%	9% - 55%	>55%
PROPORTION OF DOMINANT TAXA	<41%	41% - 71%	>71%
TOTAL EPT TAXA	>12	6 -12	<6
PROPORTION OF TOTAL EPT INDIVIDUALS	>58%	29%-58%	<29%
PROPORTION OF SHREDDERS	>6%	3%-6%	<3%

[a] Scoring criteria are based on the 1995 and 1996 reference streams.

Scoring Criteria for Channery Silt loam Region. 3rd and 4th Order Streams

Scoring Criteria for Macroinvertebrate 1B1 Metrics			
	SCORING CRITERIA[a]		
METRIC	5	3	1
TAXA RICHNESS	>20	10 - 20	<10
BIOTIC INDEX	<3.83	3.83 - 6.92	>6.92
RATIO OF SCRAPERS/ (SCRAPERS+FIL TERING COLLECTORS)	>8%	4% - 8%	<4%
PROPORTION OF HYDROPSYCHE AND CHEUMATOPSYCHE/ TOTAL EPT INDIVIDUALS	<21%	21% - 61%	>61%
PROPORTION OF DOMINANT TAXA	<48%	48% - 74%	>74%
TOTAL EPT TAXA	>12	6 - 12	<6
PROPORTION OF TOTAL EPT INDIVIDUALS	>50%	25% - 50%	<25%
PROPORTION OF SHREDDERS	>16%	8% - 16%	<8%

[a] Scoring criteria are based on the 1995 and 1996 reference streams.

Scoring Criteria for Silt Loam Region, 1st and 2nd Order Streams

Scoring Criteria for Macroinvertebrate IBI Metrics			
	SCORING CRITERIA[a]		
METRIC	5	3	1
TAXA RICHNESS	>23	12 - 23	<12
BIOTIC INDEX	<3.86	3.86 - 6.93	>6.93
RATIO OF SCRAPERS/ (SCRAPERS+FIL TERING COLLECTORS)	>20%	10% - 20%	<10%
PROPORTION OF HYDROPSYCHE AND CHEUMATOPSYCHE/ TOTAL EPT INDIVIDUALS	<15%	15% - 57%	>57%
PROPORTION OF DOMINANT TAXA	<33%	33% - 67%	>67%
TOTAL EPT TAXA	>11'	6 - 11	<6
PROPORTION OF TOTAL EPT INDIVIDUALS	>55%	28% - 55%	<28%
PROPORTION OF SHREDDERS	>5%	3% - 5%	<3%

[a] Scoring criteria are based on the 1995 and 1996 reference streams.

Scoring Criteria for Silt Loam Region, 3rd and 4th Order Streams

Scoring Criteria for Macroinvertebrate 1B1 Metrics			
	SCORING CRITERIA[a]		
METRIC	5	3	1
TAXA RICHNESS	>22	11-20	<11
BIOTIC INDEX	. <3.78	3.78 - 6.89	>6.89
RATIO OF SCRAPERS/ (SCRAPERS+FIL TERING COLLECTORS)	>18%	9% - 18%	<9%
PROPORTION OF HYDROPSYCHE AND CHEUMATOPSYCHE/ TOTAL EPT INDIVIDUALS	<17%	17% - 59%	>59%
PROPORTION OF DOMINANT TAXA	<47%	47% - 74%	>74%
TOTAL EPT TAXA	>12	7 -12	<7
PROPORTION OF TOTAL EPT INDIVIDUALS	>55%	28% - 55%	<28%
PROPORTION OF SHREDDERS	>5%	3%-5%	<3%

[a] Scoring criteria are based on the 1995 and 1996 reference streams.

APPENDIX B

STRUCTURAL STORMWATER BEST MANAGEMENT PRACTICES PERFORMANCE INDEX

STRUCTURAL STORMWATER BEST MANAGEMENT PRACTICES PERFORMANCE INDEX

PURPOSE AND BACKGROUND

The purpose of the index is to aggregate the expected effectiveness of all structural stormwater best management practices (BMPs) applied in a watershed to mitigate the full suite of negative physical and chemical effects on the ecosystem of a stream draining the catchment. The need for such an index became apparent when it was noted that the watersheds of streams in the Puget Sound region generally have more than one BMP, and often contain a number of them. The set of BMPs may be comprised of practices intended to control water quantity, water quality, or both. An index based on BMP characteristics known to influence performance allows comparing the relative levels of protection afforded different reaches of stream subject to varying stormwater management strategies and facilities. It is emphasized that the index has no intrinsic, tangible meaning, but results from a relative scoring system drawn from the literature and experience on what features of design and operation are most instrumental in performance.

GENERAL APPROACH

For a stream ecosystem reach of interest, identify its watershed and all BMPs present within it that are intended to control water quantity, water quality, or both. If a BMP provides both water quantity control and water quality control, perform the analysis for the two functions separately.

1. Assign a score for one or more measures of BMP quality that represents how close practice approaches ideal (according to the prevailing state of the art) for that measure; the score can be relative to a quantitative, physical attribute or on a semi-quantitative scale.

2. Add scores and determine what a “perfect” total score would be. Express composite scores as the sums of individual scores divided by the perfect total score.

3. Add composite scores for water quantity control and water quality control.

SPECIFIC STEPS

1. Compute composite scores for quantity control (S_{QN}; refer to SCORING WATER QUANTITY CONTROL BMPS) and quality control (S_{QL}; refer to SCORING WATER QUALITY CONTROL BMPS).

2. Add the composite quantity and quality control scores to obtain the BMP Performance Index (BMP-PI):

$$\text{BMP-PI} = S_{QN} + S_{QL}$$

3. Use the index to make relative comparisons of the anticipated effectiveness of different sets of BMPs. For example, comparisons can be made among different watersheds served by different BMPs, or before and after BMPs are installed in a watershed.

SCORING WATER QUANTITY CONTROL BMPS

Approach

This approach is based on findings in the region where the Puget Sound segment of the research project is being conducted and may not be appropriate for other areas. The Puget Sound lowlands, especially the northern and central portions, are dominated by glacial till geology in which it has been found that the prevailing soils in an undisturbed, forested condition provide 6-12 watershed-inches of water storage. Water in soil storage may eventually enter surface waters through the vadose zone or a local, and perhaps intermittent, saturated zone above the till, but at a greatly retarded rate and as baseflow augmentation. Much of the soil storage is lost in urban development through the construction of impervious surfaces and removal and compaction of soil above the till. Stormwater retention/detention aims at replacing this lost storage, but efforts to date have yielded only fractional replacement. This degree of replacement may maintain pre-development peak flows, although it often does not, but generally cannot restrict runoff volumes to pre-development levels.

With these regional observations it is proposed to set this element of the BMP Quality Index according to the estimated amount of lost soil storage replaced by structural retention/detention or nonstructural land use set aside in lieu of structural control.

Procedure

1. Identify the following areas (acres):

A_w = Total area of watershed contributing to stream segment of interest

A_d = Total developed area

A_d represents area that is actually developed. Area within an urbanized zone may be set aside or otherwise untouched by development. If soil in an area has not been disturbed by construction and vegetation is forest, pasture, meadow, etc., then do not consider this area as part of A_d. If, on

the other hand, an area has been shaped by construction equipment and planted in lawn or ornamental vegetation (including parks, cemeteries, etc.), then consider this area as part of A_d.

A_{QNdno} = Developed area with no quantity control BMPs

A_{nd} = Area not developed

Note that $A_d = A_{QNdno} + A_{QNds}$ and $A_w = A_d + A_{nd}$.

2. Score the quantity control BMPs (S_{QN}; refer to Quantity Control BMP Scoring).

Quantity Control BMP Scoring

1. Determine the lost soil storage in the developed area (LSS, acre-ft) as:

$$LSS = [A_w * DIA + 0.6 * A_d * (1 - DIA)]* 0.5 \text{ watershed-ft}$$

Where: DIA = Fraction of the developed area in impervious land cover

Refer to Attachment A for the basis of this equation.

2. Determine the U.S. Department of Agriculture Hydrologic Soil Group(s) (A, B, C, or D) on the site in question.

3. Evaluate each retention detention pond and vault and each infiltration basin and trench to estimate replacement storage as follows:

A. For infiltration basins and trenches on C soils that have been designed, built, and regularly maintained to perform infiltration and for any infiltration facilities on A or B soils even if not regularly maintained—

$$RS_I = (K * A_I * 24 \text{ hours})/(43{,}560 \text{ ft}^2/\text{acre})$$

Where: RS_I = Lost soil storage replaced by an infiltration facility (acre-ft)

K = Infiltration rate at the facility (ft/hour)

A_I = Bottom surface area of infiltration basin or side surface area of infiltration trench (ft^2)

24 hours is used because of the frequent back-to-back repetition of Pacific Northwest winter storms, which have an average duration of about 21 hours.

If the facility is constructed on D soil or on C soil and is not given regular maintenance, assume $S_I = 0$, as it is unlikely that infiltration function will be sustained long.

B. For retention/detention ponds having live storage and an outlet structure for controlled release and situated on A or B soils—

$$RS_{R/D} = (K * A_{R/D} * 6 \text{ hours})/(43{,}560 \text{ ft}^2/\text{acre}) + Z * [A_c * DIA + 0.6 * A_c * (1 - DIA)]$$

Where: $RS_{R/D}$ = Lost soil storage replaced by a infiltration from a pond (acre-ft)

$A_{R/D}$ = Bottom surface area of pond (ft^2)

6 hours is used because it is a typical period during which there is water in live storage.

Z = Fraction of lost soil storage typically compensated for by detention, depending on pond design basis [0.08 for a pond design based on the Rational Method; 0.20 for a pond designed for peak discharge rate control based on selected events and a unit hydrograph method; 0.25 for a pond designed for peak discharge rate control based on a continuous simulation method; 0.40 for a pond designed for peak discharge rate and some elevated flow duration control based on selected events and a unit hydrograph method; 0.50 for a pond designed for peak discharge rate and elevated flow duration control based on a continuous simulation method; after Booth and Jackson (1994)]

A_c = Catchment area contributing to pond (acres)

C. For retention/detention ponds having live storage and an outlet structure for controlled release and situated on C or D soils and retention/detention vaults and tanks—

$$RS_{R/D} = Z * [A_c * DIA + 0.6 * A_c * (1 - DIA)]$$

4. Determine the replacement storage (RS, acre-ft) provided by all runoff quantity control facilities serving the developed area as follows:

$$RS = \Sigma RS_{Ii} + \Sigma RS_{R/Dj}$$

Where Σ indicates summation over all BMPs

i and j signify infiltration BMPs 1, 2, 3, … n and retention/detention BMPs 1, 2, 3, … m, respectively

3. Assign as the quantity control score (S_{QN}) the ratio of total BMP storage:lost soil storage:

$$S_{QN} = RS/LSS$$

If RS is > LSS and S_{QN} is > 1, set $S_{QN} = 1$.

If quality control BMPs are also scored, add the composite quantity and quality control scores to obtain the BMP-PI:

$$\text{BMP-PI} = S_{QN} + S_{QL}$$

SCORING WATER QUALITY CONTROL BMPS

Approach

For each type of water quality control BMP find an application score as outlined below. The application score represents how well the BMP is executed (designed, built, and maintained) relative to state-of-the-art guidelines for that practice.

For each type of BMP apply an effectiveness weighting factor, which represents the pollutant removal effectiveness of that BMP relative to other practices. For most BMPs the factor is taken as the multiple of the fractional TSS reduction efficiency and the fractional total phosphorus reduction efficiency. Total phosphorus represents the relatively soluble pollutants. For oil/water separators the factor is the oil reduction efficiency. The efficiencies used are the consensus from the literature.

Procedure

1. Identify the following areas (acres):

A_w = Total area of watershed contributing to stream segment of interest

A_d = Total developed area

A_{QLdno} = Developed area with no quality control BMPs

A_{QLds} = Developed area with structural quality control BMPs

A_{nd} = Area not developed

Note that $A_d = A_{QLdno} + A_{QLds}$ and $A_w = A_d + A_{nd}$.

2. Score the structural quality control BMPs present in the developed area (refer to the sections outlining scoring for the various structural BMPs).

3. Sum the weighted condition scores for all structural water quality control BMPs serving the developed area to get S_{QL}:

$$S_{QL} = \Sigma S_{Cwk}$$

Where Σ indicates summation over all BMPs

S_{Cwi} = Weighted condition score for quality control BMP k, where k = 1, 2, 3, … p

If quantity control BMPs are also scored, add the composite quantity and quality control scores to obtain the BMP-PI:

$$\text{BMP-PI} = S_{QN} + S_{QL}$$

<u>Scoring Wet Ponds and Constructed Wetlands</u>

1. Determine volume ratio (R_{vol}) as the fraction of the wet pool storage volume relative to the volume needed to store runoff from the 6-month, 24-hour rainfall event:

A. Find wet pool volume (V_{wp}, acre-ft) from bathymetric data or by multiplying wet pool surface area times average depth.

B. Estimate the volume of runoff from the catchment served by the facility for the 6-month, 24-hour rainfall event (V_r, acre-ft).

$$V_r = C * P * A_c$$

where: C = Runoff coefficient = 0.05 + 0.009 * I

I = % of facility's catchment that is impervious

P = Rainfall depth for 6-month, 24-hour event (ft = inches/12)

A_c = Catchment area contributing to the pond or wetland (acres)

C. Compute volume ratio:

$$R_{vol} = V_{wp}/V_r$$

2. Determine geometric ratio (R_g) as the fraction of an ideal of at least five represented by the geometry:

A. Find length/width ratio (L:W)

Where: L = Length of flow path from inlet to outlet (any consistent units)

W = Average dimension perpendicular to L (any consistent units)

B. Compute geometric ratio:

$$R_g = L:W/5$$

3. Determine flow compartmentalization ratio (R_c) as the fraction of an ideal of at least two distinct cells:

If there is 1 cell only, set $R_c = 0.5$.

If there are 2 cells, set $R_c = 1.0$.

If there are 3 or more cells, set $R_c = 1.5$.

4. Determine vegetation ratio (R_{veg}) as the fraction of an ideal of at least 50% emergent herbaceous vegetation cover in the pooled storage region:

$$R_{veg} = \%\ \text{emergent herbaceous vegetation cover}/50\%$$

5. Determine maintenance ratio (R_m) as the fraction of an ideal of 3 assigned to a "very well maintained" condition:

$R_m = 3/3 = 1.0$ if very well maintained (twice/year)

$R_m = 2/3 = 0.67$ if moderately well maintained (once/year)

$R_m = 1/3 = 0.33$ if poorly maintained (< once/year)

$R_m = 0/3 = 0$ if not maintained at all

6. Determine a condition score (S_C) for the facility by summing ratios and dividing by five, which represents an ideal rating for all ratios:

$$S_C = (R_{vol} + R_g + R_c + R_{veg} + R_m)/5$$

7. Determine a weighted condition score (S_{Cw}) representing how much of the developed area the facility serves and its maximum expected pollutant removal efficiency if designed, built, and maintained according to state-of-the-art guidelines:

$$S_{Cw} = S_C * F_a * F_p$$

Where: F_a = Fraction of total developed area served by the facility = A_c/A_d

F_p = Fractional TSS reduction * Fractional TP reduction = 0.90 * 0.60 = 0.54 for wet ponds and constructed wetlands (approximate performance capability for a well designed, built, and maintained facility)

Scoring Wet Vaults

Score wet vaults in the same way as wet ponds, except set the vegetation ratio Rve = 0 and for the pollutant reduction fraction use 0.80 * 0.25 = 0.10 (approximate performance capability for a well designed, built, and maintained facility).

Scoring Extended-Detention Ponds

1. Determine dewatering time ratio (R_d) as the fraction of the actual time to drain the runoff from the 6-month, 24-hour rainfall event (t_d, hours) relative to 72 hours:

$$R_d = t_d/72$$

If there is no information about dewatering time, determine it from a standard orifice equation:

$$t_d = V_r/Q_d$$

$$Q_d = C_o * A_o * (2 * g * h)^{0.5}$$

Where: Vr = Ponded runoff volume under design event condition (ft^3)

Vr can be determined from pond geometry, either from bathymetric data or by multiplying wet pool surface area times average depth, or from the design event runoff volume:

$$V_r = C * P * A_c * 43{,}560\ ft^2/acre$$

C = Runoff coefficient = 0.05 + 0.009 * I

I = % of facility's watershed that is impervious

P = Rainfall depth for 6-month, 24-hour event (ft = inches/12)

A_c = Catchment area contributing to the pond (acres)

Q_d = Pond discharge rate (ft^3/hour = ft^3/second * 3600)

C_o = Orifice coefficient = 0.62 for circular

A_o = Area of orifice opening(s) (ft^2) = n * 3.14 * (d_o^2)/4

n = number of orifices

d_o = Diameter of orifice(s) (ft = inches/12)

g = Acceleration of gravity = 32.2 ft/$second^2$

h = Hydraulic head (ft) = Ponded depth above orifice(s) under design event condition (average if multiple orifices at different levels)

2. Determine geometric ratio (R_g). If the pond is operated in a batch mode in which it largely fills before draining begins, set R_g = 1. Otherwise, determine the ratio as the fraction of an ideal of at least five represented by the geometry:

A. Find length/width ratio (L:W)

where: L = length of flow path from inlet to outlet (any consistent units)

W = average dimension perpendicular to L (any consistent units).

B. Compute geometric ratio:

$$R_g = L:W/5$$

3. Determine flow compartmentalization ratio (R_c) as the fraction of an ideal of at least two distinct cells:

If there is 1 cell only, set R_c = 0.5.

If there are 2 cells, set R_c = 1.0.

If there are 3 or more cells, set R_c = 1.5.

4. Determine maintenance ratio (R_m) as the fraction of an ideal of 3 assigned to a "very well maintained" condition:

R_m = 3/3 = 1.0 if very well maintained (twice/year)

R_m = 2/3 = 0.67 if moderately well maintained (once/year)

R_m = 1/3 = 0.33 if poorly maintained (< once/year)

R_m = 0/3 = 0 if not maintained at all

5. Determine a condition score (S_C) for the facility by summing ratios and dividing by four, which represents an ideal rating for all ratios:

$$S_C = (R_d + R_g + R_c + R_m)/4$$

6. Determine a weighted condition score (S_{Cw}) representing how much of the developed area the facility serves and its maximum expected pollutant removal efficiency if designed, built, and maintained according to state-of-the-art guidelines:

$$S_{Cw} = S_C * F_a * F_p$$

Where: F_a = Fraction of total developed area served by the facility = A_c/A_d

F_p = Fractional TSS reduction * Fractional TP reduction = 0.70 * 0.20 = 0.14 for extended-detention ponds (approximate performance capability for a well designed, built, and maintained facility)

Scoring extended-detention vaults

Score extended-detention vaults the same as extended-detention ponds, except for the pollutant reduction fraction use 0.70 * 0.10 = 0.07 (approximate performance capability for a well designed, built, and maintained facility).

Scoring Biofiltration Swales and Filter Strips

1. Determine capacity ratio (R_c) as the fraction of the swale treatment volume or filter strip treatment surface area provided relative to the quantity needed to provide sufficient residence time to runoff from the 6-month, 24-hour rainfall event for common conditions:

Swale—

$$R_c = (A_s * L_s)/V_{est}$$

Where: A_s = Swale cross-sectional area (ft^2) ≈ 0.17 ft maximum flow depth * swale width

L_s = Swale length (ft)

V_{est} = Estimated necessary volume (ft^3) ≈ 2.5 * I * A_c + 18 [Note: This regression equation was derived in a theoretical analysis by Horner (1996) and explains 99% of the variance in V_{est}.]

I = % of facility's watershed that is impervious

A_c = Catchment area contributing to the swale (acres)

Filter strip—

$$R_c = (W_{fs} * L_{fs})/A_{est}$$

Where: W_{fs} = Filter strip width (perpendicular to flow, ft)

L_{fs} = Filter strip length (parallel to flow, ft)

A_{est} = Estimated necessary surface area (ft^2) ≈ 0.30 * W_c + 0.52 * s – 1.5 [Note: This multiple regression equation explains 93 percent of the variance in A_{est} and was based on analysis of a design procedure by King County Surface Water Management Division (1998).]

W_c = Width of runoff contributing area (perpendicular to flow, ft)

s = Filter strip slope (%)

2. Determine slope ratio (R_s):

Swale—

If the longitudinal slope is 1.5-2.5%, 2.5-6.0% with check dams for every 1 ft of fall, or < 1.5 % with underdrains or wetland plants, set R_s = 1.0.

Otherwise, set R_s = 0.

Filter strip—

If the longitudinal slope is < 15%, set R_s = 1.0.

If the longitudinal slope is 15-20%, set R_s = 0.50.

Otherwise, set R_s = 0.

3. Determine flow introduction ratio (R_i):

Swale--

If there are features to distribute influent across the swale, such as a good level spreader, perforated pipe, or stilling well; and if there are features to dissipate energy, such as rip-rap or a stilling well, set R_i = 1.0.

Otherwise, set R_i = 0.50.

Filter strip--

If there is no evidence that runoff concentrates anywhere along the width of the strip, set R_i = 1.0.

Otherwise, set R_i = 0.50.

4. Determine maintenance ratio (R_m) as the fraction of an ideal of 3 assigned to a "very well maintained" condition:

R_m = 3/3 = 1.0 if very well maintained (must have nearly complete cover of dense, fine vegetation [95-100%], as well as other evidence of good maintenance)

R_m = 2/3 = 0.67 if moderately well maintained (assign if relatively minor vegetation gaps [70-95% cover] and other maintenance moderately good)

R_m = 1/3 = 0.33 if poorly maintained (assign if significant vegetation gaps [40-70% cover], regardless of other maintenance)

R_m = 0/3 = 0 if not maintained at all (assign if very significant vegetation gaps [<40% cover], regardless of other maintenance)

5. Determine a condition score (S_C) for the facility by summing ratios and dividing by four, which represents an ideal rating for all ratios:

$$S_C = (R_c + R_s + R_i + R_m)/4$$

6. Determine a weighted condition score (S_{Cw}) representing how much of the developed area the facility serves and its maximum expected pollutant removal efficiency if designed, built, and maintained according to state-of-the-art guidelines:

$$S_{Cw} = S_C * F_a * F_p$$

Where: F_a = Fraction of total developed area served by the facility = A_c/A_d

F_p = Fractional TSS reduction * Fractional TP reduction = 0.80 * 0.30 = 0.24 for biofiltration swales and filter strips (approximate performance capability for a well designed, built, and maintained facility)

Scoring Infiltration Facilities (Basins, Trenches, Etc.)

1. Determine infiltration volume ratio (R_{iv}) as the fraction of the volume infiltrated in 24 hours (used because of the frequent back-to-back repetition of Pacific Northwest winter storms) relative to the runoff volume from the 6-month, 24-hour rainfall event:

A. Find volume infiltrated in 24 hours (V_i, acre-ft):

$$V_i = (K * A_i * 24 \text{ hours})/(43{,}560 \text{ ft}^2/\text{acre})$$

Where: K = Infiltration rate (ft/hour)

A_i = Bottom surface area of infiltration facility (ft^2)

If the facility is constructed in Hydrologic Soil Group D soil, assume K = 0, as any infiltration function will not be sustained long. Also assume K = 0 in Hydrologic Soil Group C soil, unless the facility has been designed and built and is regularly maintained to perform infiltration.

B. Find the volume of runoff from the watershed served by the facility for the 6-month, 24-hour rainfall event (V_r, acre-ft):

$$V_r = C * P * A_c$$

where: C = Runoff coefficient = 0.05 + 0.009 * I

I = % of watershed that is impervious

P = Rainfall depth for 6-month, 24-hour event (ft = inches/12)

A_c = Catchment area contributing to the infiltration facility (acres)

C. Compute infiltration volume ratio:

$$R_{iv} = V_i/V_r$$

2. Determine maintenance ratio (R_m) as the fraction of an ideal of 3 assigned to a "very well maintained" condition:

R_m = 3/3 = 1.0 if very well maintained (must be no evidence of any restriction of infiltration, as well as other evidence of good maintenance)

R_m = 2/3 = 0.67 if moderately well maintained (assign if relatively minor evidence of restriction of infiltration and other maintenance moderately good)

R_m = 1/3 = 0.33 if poorly maintained (assign if significant restriction of infiltration, regardless of other maintenance)

R_m = 0/3 = 0 if not maintained at all (assign if very significant restriction of infiltration, regardless of other maintenance)

3. Determine a condition score (S_C) for the facility by summing ratios and dividing by two, which represents an ideal rating for all ratios:

$$S_C = (R_{iv} + R_m)/2$$

4. Determine a weighted condition score (S_{Cw}) representing how much of the developed area the facility serves and its maximum expected pollutant removal efficiency if designed, built, and maintained according to state-of-the-art guidelines:

$$S_{Cw} = S_C * F_a * F_p$$

Where: F_a = Fraction of total developed area served by the facility = A_c/A_d

F_p = Fractional TSS reduction * Fractional TP reduction = 1.0 * 1.0 = 1.0 for infiltration facilities (approximate performance capability for a well designed, built, and maintained facility)

<u>Scoring And Filters</u>

1. Determine filtration volume ratio (R_{fv}) as the fraction of the actual filter volume (V_{fa}, ft^3) relative to the design filter volume (V_{fd}, ft^3) recommended for the particular filter type:

$$R_{fv} = V_{fa}/V_{fd}$$

Delaware-type sand filter—V_{fd} = 540 ft^3 per acre of catchment contributing runoff (acres) [Note: V_{fd} represents the volume of the sand chamber, with the sedimentation chamber to have equivalent volume.]

Austin-type sand filter—V_{fd} = Volume of runoff from the catchment served by the facility from the first 0.5 inch of rainfall [Note: V_{fd} represents the total volume of sand filter and sedimentation chambers, with the latter at least 20% of the total.]

$$V_{fd} = 3630 * C * P * A_c$$

Where: C = Runoff coefficient = 0.05 + 0.009 * I

I = % of watershed that is impervious

$P = 0.5$ inch

A_c = Catchment area contributing to the sand filter (acres)

2. Determine maintenance ratio (R_m) as the fraction of an ideal of 3 assigned to a "very well maintained" condition:

$R_m = 3/3 = 1.0$ if very well maintained (twice/year)

$R_m = 2/3 = 0.67$ if moderately well maintained (once/year)

$R_m = 1/3 = 0.33$ if poorly maintained (< once/year)

$R_m = 0/3 = 0$ if not maintained at all

3. Determine a condition score (S_C) for the facility by summing ratios and dividing by two, which represents an ideal rating for all ratios:

$$S_C = (f_{iv} + R_m)/2$$

4. Determine a weighted condition score (S_{Cw}) representing how much of the developed area the facility serves and its maximum expected pollutant removal efficiency if designed, built, and maintained according to state-of-the-art guidelines:

$$S_{Cw} = S_C * F_a * F_p$$

Where: F_a = Fraction of total developed area served by the facility = A_c/A_d

F_p = Fractional TSS reduction * Fractional TP reduction (approximate performance capability for a well designed, built, and maintained facility)

Delaware-type sand filter--Fp = 0.80 * 0.40 = 0.32

Austin-type sand filter--Fp = 0.80 * 0.40 = 0.32

ATTACHMENT A

BASIS FOR LOST SOIL STORAGE EQUATION (QUANTITY CONTROL BMP SCORING STEP 1)

The equation is:

$$LSS = [A_w * DIA + 0.6 * A_d * (1 - DIA)]* 0.5 \text{ watershed-ft}$$

The equation would typically apply to a catchment on glacial till with single-family residential land use. Burges, Wigmosta, and Meena (1998) measured runoff from such a catchment in the Puget Sound Lowlands having 30 percent impervious area and the balance almost entirely in suburban lawn. They determined that the pervious 70 percent produced 60 percent of the annual runoff volume. The runoff coefficient C_p for that portion can be estimated as follows:

$0.7 * C_p = 0.6 * V$, where V = annual runoff volume

$V = P * (C_p * A_p + C_i + A_i)$, where P = precipitation, A_p = pervious area, C_i = impervious area runoff coefficient, A_i = impervious area

C_i is typically approximately 0.9; for unit P; then, $V = C_p * 0.7 + 0.9 * 0.3$

The two equations in two unknowns V and C_p can be solved for C_p to obtain $C_p = 0.57$. This value was rounded to 0.6 for the lost soil storage equation.

The lost soil storage equation assumes that all storage is lost in impervious areas, represented by the first term within the brackets, and 60 percent is lost in pervious areas, represented by the second term. According to Booth (1991) and Booth (personal communication), forested Puget Sound Lowland catchments provide 6-12 watershed-inches of soil storage. A value at the bottom of the recognized range of presumed lost soil storage (6 watershed-inches = 0.5 watershed-ft) was selected in recognition that developed land uses provide some storage, and disturbed, undeveloped land does not provide as much storage as forest.

ATTACHMENT B

REFERENCES

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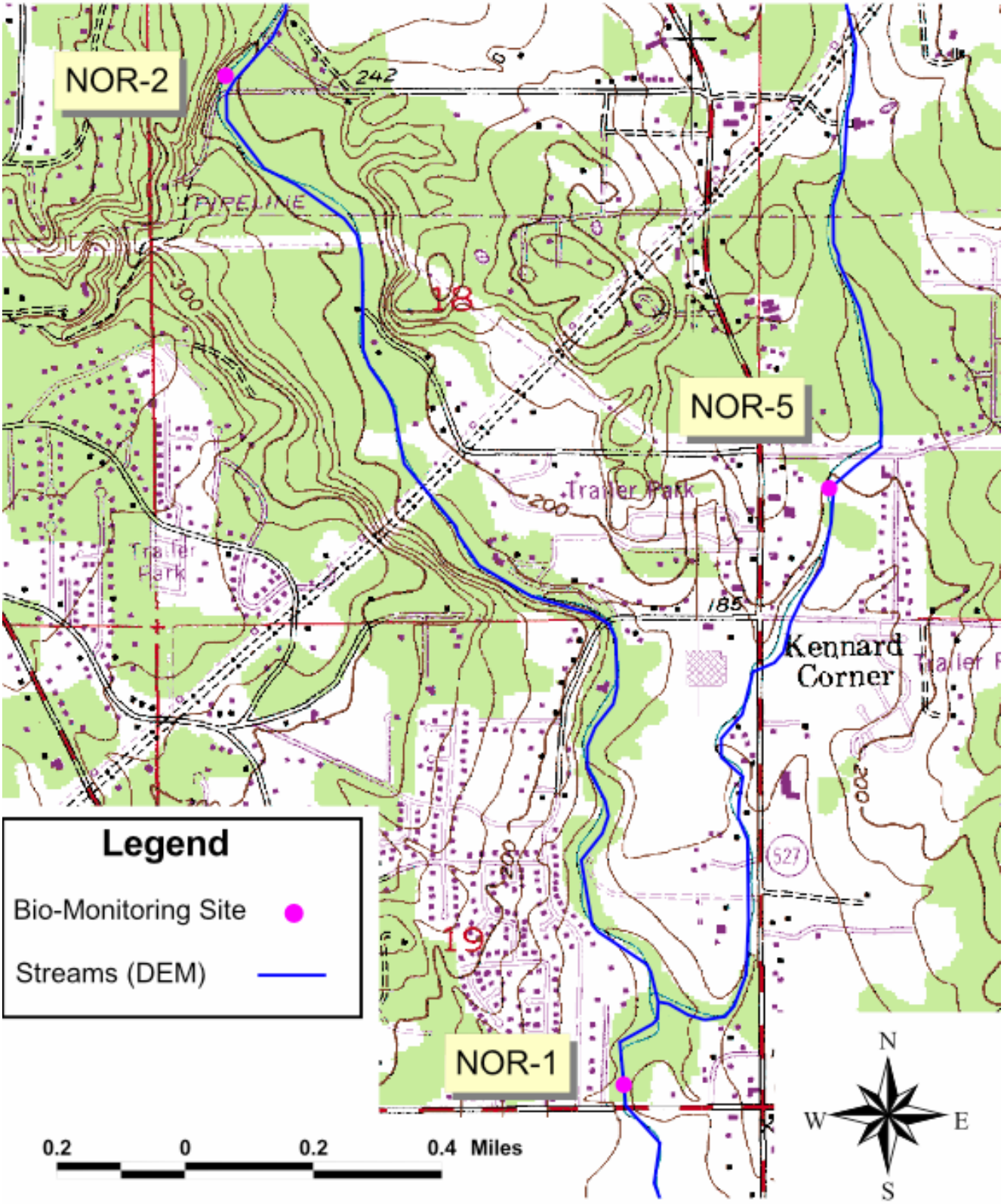
King County Surface Water Management Division. 1998. Surface Water Design Manual. King County Department of Public Works, Seattle, WA.

APPENDIX C

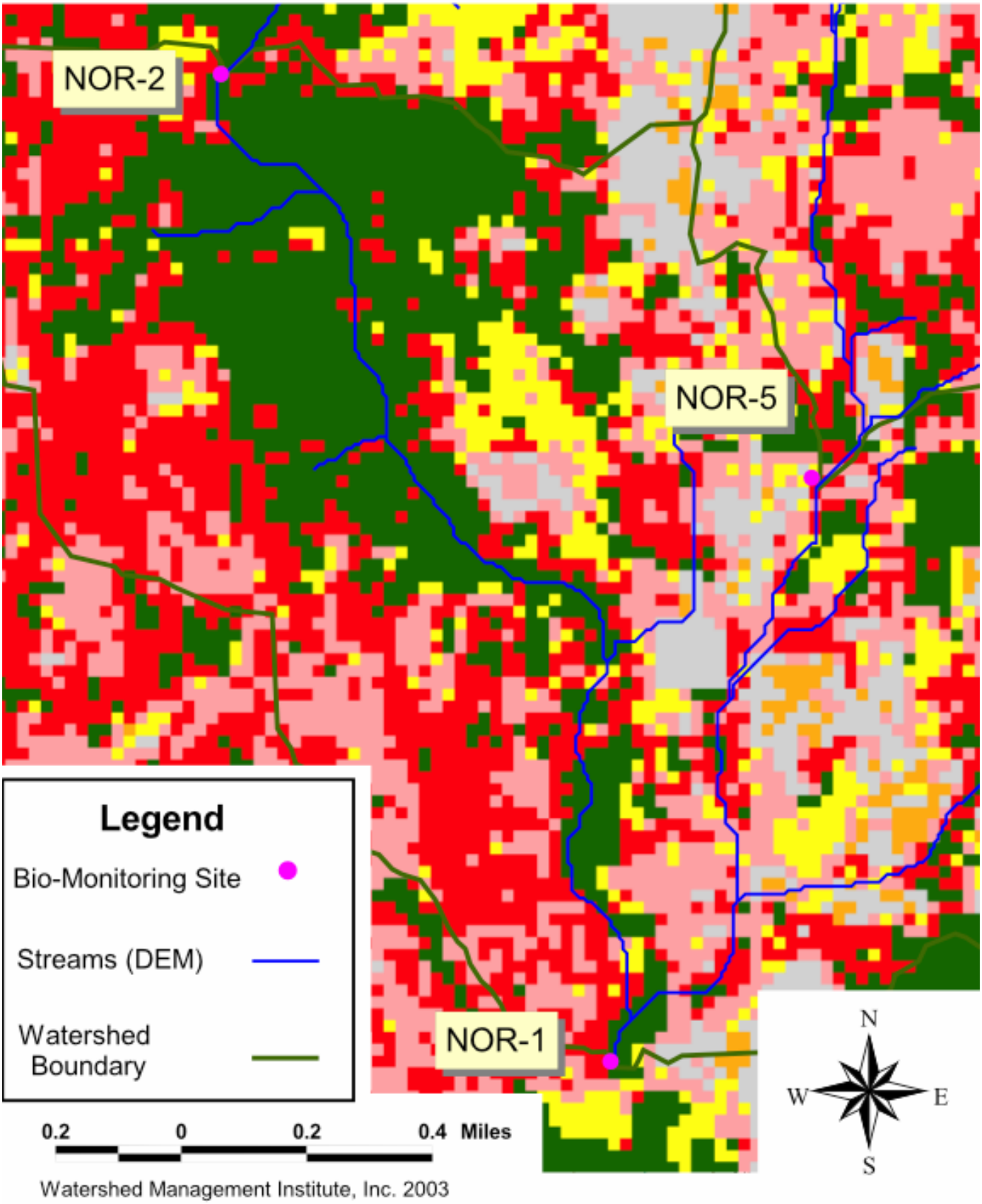
EXAMPLE MAPS FOR GIS-BASED WATERSHED ASSESSMENT

Site Locations

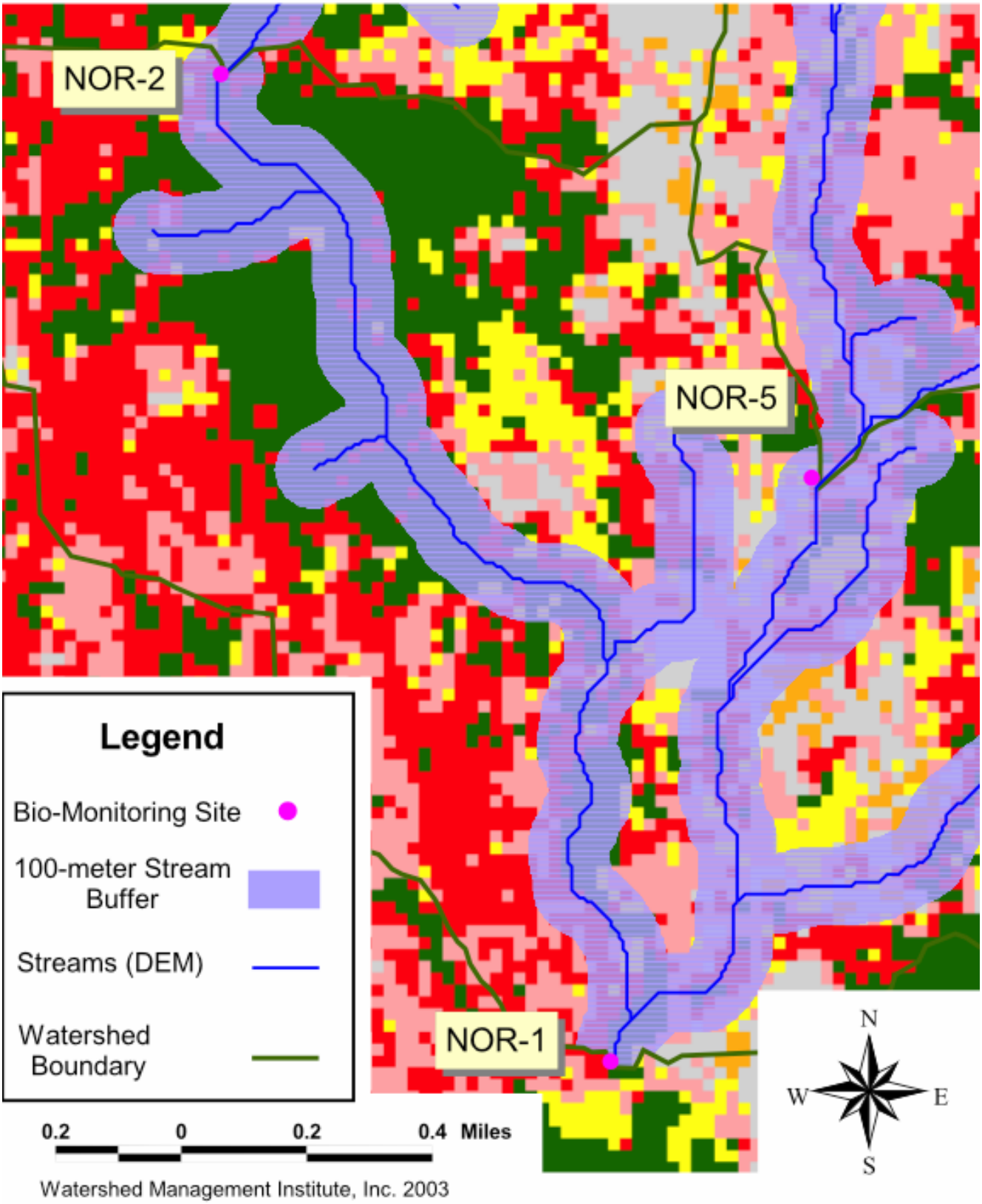


Watershed Management Institute, Inc. 2003

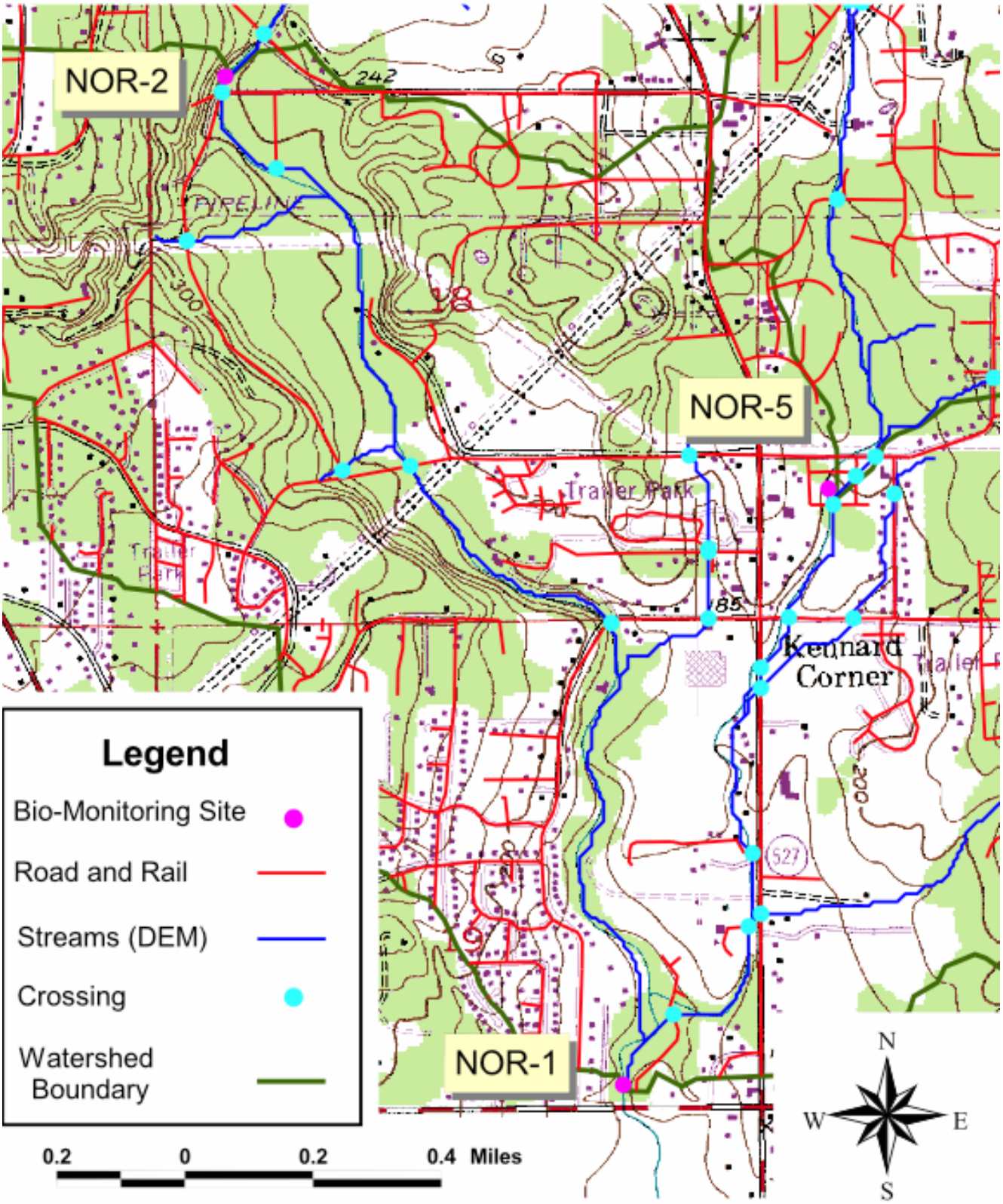
Watershed LU/LC



Stream Riparian

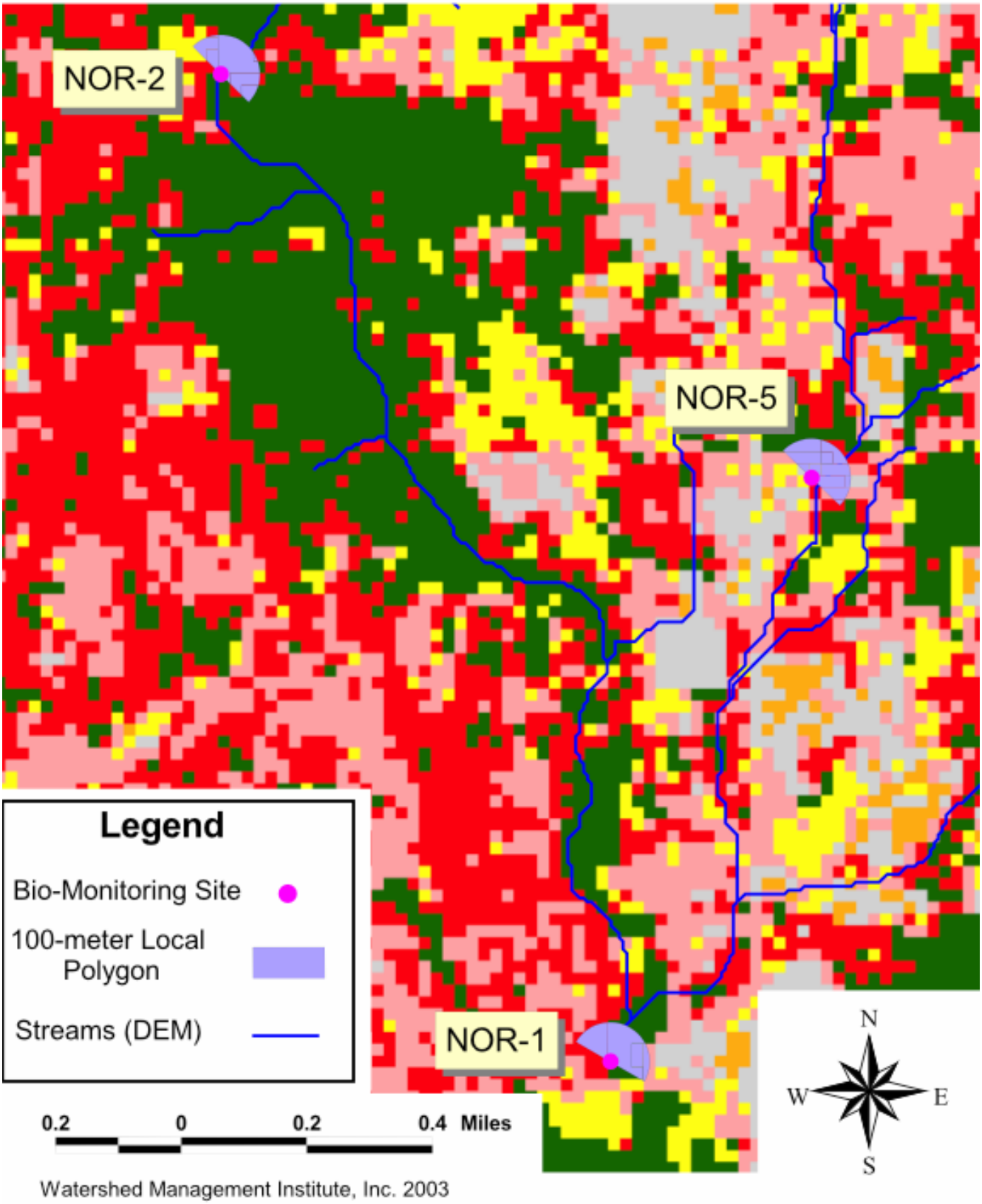


Riparian Fragmentation



Watershed Management Institute, Inc. 2003

Local LU/LC Example



APPENDIX D

WATERSHED ANALYSIS MATRIX

Watershed Scale	GIS Metric	Metric Objective	Ecological Basis	GIS Notes	Montgomery County	Austin	Puget Sound
1	% Native Forest Cover	Measure of Natural Land-Cover	Hydrology & Habitat		X	X	X
2	% Mature Forest Cover	Measure of Natural Land-Cover	Hydrology & Habitat		X		
3	% Young Forest Cover	Measure of Natural Land-Cover	Hydrology & Habitat		X		
4	% Coniferous Forest Cover	Measure of Natural Land-Cover	Hydrology & Habitat	>60% C	X	X	
5	% Deciduous Forest Cover	Measure of Natural Land-Cover	Hydrology & Habitat	>60% D	X	X	
6	% Mixed Forest Cover	Measure of Natural Land-Cover	Hydrology & Habitat	<60% C&D			
7	% Natural Wetlands	Measure of Natural Land-Cover	Hydrology & Habitat	NWI	X	X	X
8	% Native Shrub Cover	Measure of Natural Land-Cover	Hydrology & Habitat		X	X	X
9	% Native Grassland or Prairie	Measure of Natural Land-Cover	Hydrology & Habitat			X	
10	% Natural Land-Cover	Measure of Natural Land-Cover	Hydrology & Habitat	Sum (1-8)	X	X	X
11	% Turf-Grass (Lawns, Parks, Golf, & Sports Fields)	Measure of Modified Land-Cover	Hydrology & Habitat		X	X	X
12	% Invasive & Exotic Species	Measure of Modified Land-Cover					
13	Mean Forest Patch Size	Measure of Modified Land-Cover	Forest Fragmentation	Frag-Stats			
14	Maximum Forest Patch Size	Measure of Modified Land-Cover	Forest Fragmentation	Frag-Stats			
15	Minimum Forest Patch Size	Measure of Modified Land-Cover	Forest Fragmentation	Frag-Stats			
16	Forest Patch Size Evenness/Diversity	Measure of Modified Land-Cover	Forest Fragmentation	Frag-Stats			
17	Forest Patch Connectivity	Measure of Modified Land-Cover	Forest Fragmentation	Frag-Stats			
18	% Developed	Measure of Modified Land-Cover	Hydrology & WQ	Reciprocal of #10		X	

Watershed Scale	GIS Metric	Metric Objective	Ecological Basis	GIS Notes	Montgomery County	Austin	Puget Sound
19	% Clear-Cut or Timber Harvest	Measure of Modified Land-Cover	Hydrology & WQ	Stand-Age			
20	% Agricultural Land-Use	Agricultural & Livestock Grazing Impact	Hydrology & WQ		X	X	X
21	% Institutional	Measure of specific type of Land-Use	Hydrology & WQ	Schools, Civic, & Gov't		X	
22	% Industrial	Measure of specific type of Land-Use	Hydrology & WQ		X	X	
23	% Commercial	Measure of specific type of Land-Use	Hydrology & WQ	Strip-Malls & Shopping Centers	X	X	
24	% Office-Park	Measure of specific type of Land-Use	Hydrology & WQ	Business & Medical		X	
25	% Urban SF Residential	Measure of specific type of Land-Use	Hydrology & WQ		X		
26	% Multi-Family	Measure of specific type of Land-Use	Hydrology & WQ		X	X	
27	% Suburban SF Residential	Measure of specific type of Land-Use	Hydrology & WQ		X		
28	% HD Suburban SF Residential	Measure of specific type of Land-Use	Hydrology & WQ			X	
29	% MD Suburban SF Residential	Measure of specific type of Land-Use	Hydrology & WQ			X	
30	% LD Suburban SF Residential	Measure of specific type of Land-Use	Hydrology & WQ				
31	% Rural SF Residential	Measure of specific type of Land-Use	Hydrology & WQ		X	X	
32	% Transportation & Utilities	Measure of specific type of Land-Use	Hydrology & WQ		X	X	
33	% Paved-Urban	Measure of specific type of Land-Use	Hydrology & WQ				X
34	% Grass-Urban	Measure of specific type of Land-Use	Hydrology & WQ				X
35	% Forest-Urban	Measure of specific type of Land-Use	Hydrology & WQ				X

Watershed Scale	GIS Metric	Metric Objective	Ecological Basis	GIS Notes	Montgomery County	Austin	Puget Sound
36	%TIA	Total Impervious Surface Area	Hydrology & WQ	Regional LULC Conversion Factors	X	X	X
37	%EIA	Effective Impervious Surface Area	Hydrology & WQ	Used for Hydrologic Modeling			
38	Mean Basin Slope	Average Watershed Slope	Runoff & Erosion	Used for Hydrologic Modeling			
39	Soil Type	Distribution of soils in the watershed	Runoff, Erosion, & Infiltration	NRCS Soil Survey Data			
40	Dominant Land-Use	Measure of Dominant Land-Use	Type of Human Influence	20-35			
41	Road-Density	(Total Road Length) / (Basin Area)	Watershed Fragmentation	DOT Data			
42	Highway Density	(Total Highway Length) / (Basin Area)	Vehicle Related WQ Factors	DOT Data			
43	ADT	Average Daily Traffic	Vehicle Related WQ Factors	DOT Data			
44	Stormwater Piping Density	Length of Stormwater Piping per Basin Area	Hydrology & WQ	PW Data			
45	Stormwater Outfall Frequency	Number of Outfalls per Basin Area	Hydrology & WQ	PW Data			
46	Stormwater BMP Coverage	Number, Volume, & Quality of BMPs	Hydrology & WQ	PW Data			
47	Stormwater BMP Density	(BMP Treatment Area) / (Basin Area)	Hydrology & WQ	PW Data			
48	Hydrologic Runoff Potential	(Basin Area) * (Mean Annual Rainfall)	Runoff Potential				
49	% Steep Slopes	% Slopes > 30%	Erosion & Mass-Wasting				
50	Watershed Population	General Measure of Human Influence	Level of Human Influence				

Riparian Scale	GIS Metric	Metric Objective	Ecological Basis	GIS Notes	Montgomery County	Austin	Puget Sound
1	% Natural Land-Cover w/in 300 m	Measure of Buffer Width	Lateral Riparian Extent		X	X	X
2	% Natural Land-Cover w/in 100 m	Measure of Buffer Width	Lateral Riparian Extent		X	X	X
3	% Natural Land-Cover w/in 50 m	Measure of Buffer Width	Lateral Riparian Extent		X	X	X
4	% Natural Land-Cover w/in 30 m	Measure of Buffer Width	Lateral Riparian Extent		X	X	
5	% Natural Land-Cover w/in 10 m	Measure of Buffer Width	Lateral Riparian Extent			X	
6	% Each Land-Cover Type w/in 300 m	Measure of Riparian Quality	Habitat, Hydrology, & WQ	Regional LULC	X	X	X
7	% Each Land-Cover Type w/in 100 m	Measure of Riparian Quality	Habitat, Hydrology, & WQ	Regional LULC	X	X	X
8	% Each Land-Cover Type w/in 50 m	Measure of Riparian Quality	Habitat, Hydrology, & WQ	Regional LULC	X	X	X
9	% Each Land-Cover Type w/in 30 m	Measure of Riparian Quality	Habitat, Hydrology, & WQ	Regional LULC	X	X	
10	% Each Land-Cover Type w/in 10 m	Measure of Riparian Encroachment	Habitat, Hydrology, & WQ	Regional LULC		X	
11	% Each Land-Use Type w/in 300 m	Measure of Riparian Encroachment	Habitat, Hydrology, & WQ	Regional LULC	X	X	X
12	% Each Land- Use Type w/in 100 m	Measure of Riparian Encroachment	Habitat, Hydrology, & WQ	Regional LULC	X	X	X
13	% Each Land- Use Type w/in 50 m	Measure of Riparian Encroachment	Habitat, Hydrology, & WQ	Regional LULC	X	X	X
14	% Each Land- Use Type w/in 30 m	Measure of Riparian Encroachment	Habitat, Hydrology, & WQ	Regional LULC	X	X	
15	% Each Land- Use Type w/in 10 m	Measure of Riparian Encroachment	Habitat, Hydrology, & WQ	Regional LULC		X	

Riparian Scale	GIS Metric	Metric Objective	Ecological Basis	GIS Notes	Montgomery County	Austin	Puget Sound
16	%TIA w/in 300 m	Measure of Riparian Quality	Hydrology & WQ		X	X	X
17	%TIA w/in 100 m	Measure of Riparian Quality	Hydrology & WQ		X	X	X
18	%TIA w/in 50 m	Measure of Riparian Quality	Hydrology & WQ		X	X	X
19	%TIA w/in 30 m	Measure of Riparian Quality	Hydrology & WQ		X	X	
20	%TIA w/in 10 m	Measure of Riparian Quality	Hydrology & WQ			X	
21	% Floodplain Modification	% Floodplain Area Developed	Hydrology & WQ				
22	% Stream Channel Modification	% Armored or Channelized	Habitat & Hydrology				
23	% Stream Piping	% Stream w/in Piping	Habitat & Hydrology				
24	Stormwater Outfall Frequency	# Outfalls per Stream Channel Length	Hydrology & WQ	(#per km)			
25	Riparian Fragmentation	# Breaks in Riparian Corridor Normalized to Stream Length	Longitudinal Riparian Connectivity	(#per km)	X	X	X

Local Scale	GIS Metric	Metric Objective	Ecological Basis	GIS Notes	Montgomery County	Austin	Puget Sound
1	% Each Land-Cover Type w/in LIZ[a]	Measure of Local Conditions at Sample or Survey Site	Habitat, Hydrology, & WQ	30, 50, & 100 m Arc	X	X	X
2	% Each Land-Use Type w/in LIZ[a]	Measure of Local Conditions at Sample or Survey Site	Habitat, Hydrology, & WQ	30, 50, & 100 m Arc	X	X	X
3	%TIA w/in LIZ[a]	Measure of Local Conditions at Sample or Survey Site	Habitat, Hydrology, & WQ	30, 50, & 100 m Arc	X	X	X
4	Mean or Median Distance to Nearest Development	Linear Distance to Nearest Structure	Hydrology, & WQ				
5	Mean or Median Distance to Nearest Road	Linear Distance to Nearest Road	Hydrology, & WQ				
6	Mean or Median Distance from Nearest Stormwater Outfall	Linear Distance to Nearest Stormwater Outfall	Hydrology, & WQ				
7	% Forest w/in 100 m Riparian Buffer	Local Riparian Extent & Quality	Habitat, Hydrology, & WQ	100 m upstream only		X	
8	% Forest w/in 50 m Riparian Buffer	Local Riparian Extent & Quality	Habitat, Hydrology, & WQ	100 m upstream only			
9	% Forest w/in 30 m Riparian Buffer	Local Riparian Extent & Quality	Habitat, Hydrology, & WQ	100 m upstream only			
10	% Forest w/in 10 m Riparian Buffer	Local Riparian Extent & Quality	Habitat, Hydrology, & WQ	100 m upstream only			

[a] LIZ = *local influence zone* or "arc of human influence". It is important to analyze for a local influence zone that is meaningful from both an ecological and anthropogenic perspective for the region in which the study stream is located. LIZ can be defined in various ways relative to the stream sampling location, for example, as 300-meter diameter zone extending upstream of the sampling location or a rectangular zone 100 meters on each side of the stream extending 1 km upstream. The purpose of this "local" analysis is to identify any significant localized contributions to the cumulative impacts measured at the sample site.

APPENDIX E

SCORING OF WATERSHED CONDITION AND HABITAT QUALITY METRICS

SCORING OF PUGET SOUND WATERSHED CONDITION INDEX METRICS
(All values are in terms of percent.)

METRIC	1	2	3	4	5
Watershed forest	< 15	15-29	30-49	50-89	≥ 90
Watershed TIA	> 45	35-45	12-34	5-11	< 5
300-m buffer TIA	> 45	40-44	25-39	15-24	< 15
300-m buffer forest	< 15	15-29	30-59	60-69	≥ 70
50-m buffer TIA	> 45	35-45	25-34	10-24	< 10
50-m buffer forest	< 15	15-29	30-44	45-74	≥ 75
300-m local paved + urban grass-shrub	> 55	45-55	25-44	10-24	< 10

SCORING OF AUSTIN WATERSHED CONDITION INDEX METRICS
(All values are in terms of percent.)

METRIC	1	2	3	4	5
100-m buffer transport	> 35	16-35	11-15	6-10	≤ 5
Watershed TIA	> 55	31-55	21-30	6-20	≤ 5
Watershed transport	> 25	21-25	16-20	6-15	≤ 5
10-m buffer TIA	> 55	41-55	21-40	6-20	≤ 5
100-m buffer TIA	> 55	36-55	16-35	6-15	≤ 5
Local natural land cover	< 40	41-60	61-75	76-85	> 85

SCORING OF MONTOGOMERY COUNTY WATERSHED CONDITION INDEX METRICS
(All values are in terms of percent.)

METRIC	1	2	3	4	5
Watershed roads	> 10	8-10	6-7	3-5	≤ 2
Watershed roofs	> 12	11-12	7-10	5-6	≤ 4
Watershed TIA	> 35	31-35	26-30	21-25	≤ 20
Watershed parking	> 12	10-12	6-9	4-5	≤ 3
Watershed native forest	< 40	NA[a]	41-65	NA[a]	> 65

[a] NA—Not assigned

SCORING OF PUGET SOUND HABITAT QUALITY INDEX METRICS

METRIC	1	2	3	4
Glide habitat (%)	≥ 70	60-69	40-59	< 40
Substrate embeddedness (%)	≥ 50	40-49	30-39	< 30
Pool frequency (no./km)	< 40	40-64	65-79	≥ 80
Large woody debris frequency (no./km)	< 75	75-149	150-224	≥ 225
Large woody debris frequency (no./bank-full width)	< 1.0	1.0-1.4	1.5-1.9	≥ 2.0
Large woody debris volume (m^3/km)	< 100	101-299	300-399	≥ 400
Streambank stability rating (no unit)	1	2	3	4