United States
Environmental Protection
Agency

Clinch and Powell Valley Watershed Ecological Risk Assessment

EPA/600/R-01/050
September 2002

Clinch and Powell Valley Watershed Ecological Risk Assessment

National Center for Environmental Assessment
Office of Research and Development
U.S. Environmental Protection Agency
Washington, DC

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ABSTRACT

A watershed ecological risk assessment of the unique Clinch and Powell river system in southwestern Virginia strongly suggests that (1) coal mining activities and agricultural practices, past and present, are having adverse impacts on stream habitats, resulting in unacceptable losses of valuable and rare native fish and mussels and (2) prompt implementation of practical risk-lowering actions, such as reclaiming abandoned mines, spill prevention, excluding livestock from streams, and establishing riparian vegetation zones, can mitigate these adverse effects in the future.

The free-flowing Clinch and Powell Valley watershed, which drains into Norris Lake in northeastern Tennessee, has historically had one of the richest assemblages of native fish and freshwater mussels in the world. Nearly half of the species historically present are now extinct, threatened, or endangered. The U.S. Environmental Protection Agency's ecological risk assessment framework was used to structure a watershed-scale analysis of associations between land use and in-stream habitat and their effects on fish and mussels.

A pilot study of one of four subwatersheds determined that the fish Index of Biotic Integrity (IBI) was a useful surrogate for mussel species richness and found the optimal spatial scale to describe associations between land use, stressors, and biota. These findings were used to structure the watershed risk analysis of relationships between sources, stressors, and effects.

Percent pasture area, percent crop land, and proximity to active mining, urban areas, or major transportation routes accounted for more than half of the variance in fish IBI scores, with coal mining having the most impact. Native fish and mussel populations appeared to be at greatest risk as more stressors co-occurred. Our results indicate that a number of sources and stressors are responsible for the decline in native species in the Clinch and Powell Valley watershed, but naturally vegetated riparian corridors may help mitigate some of these effects.

Preferred citation:
U.S. Environmental Protection Agency (EPA) (2002). Clinch and Powell Valley watershed ecological risk assessment. National Center for Environmental Assessment, Washington, DC; EPA/600/R-01/050. Available from: National Technical Information Service, Springfield, VA. <http://www.epa.gov/ncea>

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LIST OF ACRONYMS

BMP	Best Management Practices
CMCP	Cuberlandian Mollusc Conservation Program
CPRATS	Clinch-Powell River Action Team Survey
EPA	United States Environmental Protection Agency
EPT	*Ephemoptera, Plecoptera, Trichoptera*
FWS	United States Fish and Wildlife Service
GIS	Geographic Information System
IBI	Index of Biotic Integrity
NCEA–W	National Center for Environmental Assessment–Washington, DC
SAMAB	Southern Appalachian Man and the Biosphere
TNC	The Nature Conservancy
TVA	Tennessee Valley Authority
USGS	United States Geological Survey
VDEQ	Virginia Department of Environmental Quality
VDOT	Virginia Department of Transportation

GLOSSARY

Allochthonous Energy: energy created outside the ecosystem. Commonly used to refer to organic matter produced from photosynthesis in the watershed rather than within the waterbody.

Assessment Endpoint: an explicit expression of the environmental value that is to be protected, operationally defined by an ecological entity and its attributes.

Benthic: bottom-dwelling.

Best Management Practices (BMPs): methods that have been determined to be the most effective, practical means of preventing or reducing pollution from nonpoint sources.

Biomagnification: the increased accumulation and concentration of a contaminant at higher levels of the food web because the contaminants are not broken down within organisms.

Detritus: particles of dead and decaying organic matter.

Embeddedness: the extent to which rocks (gravel, cobble, and boulders) are surrounded by, covered, or sunken into the silt, sand, or mud of the stream bottom. Generally, as rocks become embedded, fewer living spaces are available to macroinvertebrates and fish for shelter, spawning, and egg incubation.

Endemic: native to a particular region.

Ephemeroptera, Plecoptera, Trichoptera (EPT): a measure of the quality of the macroinvertebrate community, based on the number of species of macroinvertebrates found in three taxonomic families Ephemeroptera, Plecoptera, Trichoptera.

Extirpation: small-scale eradication of a species.

Extrapolate: to infer or estimate by extending or projecting known information.

Fines: fine particulate matter (e.g., clay, silt).

Fish Index of Biotic Integrity (IBI): a series of measures that describe the quality of the fish community, based on characteristics of individual fish (e.g., presence of fish tumors), fish populations (e.g., percent juveniles of a given species), and the composition of the fish community (presence of pollution-tolerant species).

Fragmentation: the process of transforming large continuous forest patches into one or more smaller patches surrounded by disturbed areas.

Glochidia: an obligate parasitic larval stage of the mussel that must attach onto the fins, epidermis, or gills of a suitable host fish.

Instream Cover: area available to aquatic biota for protection, shelter, spawning, and feeding.

Karst: a terrain generally underlain by limestone or dolomite in which the topography is generally formed by dissolving of rock and that may be characterized by sinkholes, sinking streams, closed depressions, subterranean drainage, and caves.

Measure of Effect: a change in an attribute of an assessment endpoint or its surrogate in response to a stressor.

Metrics: ecologically relevant measures of assemblage attributes used to analyze changes due to stressors.

Recruitment: the addition of new individuals to the existing population.

Refugia: areas that provide organisms with protection from predators, storms, etc.

Riparian: an area that borders a waterbody and serves as a transition zone between aquatic ecosystems and terrestrial ecosystems.

Riparian Buffer: the width of the streamside vegetated area perpendicular to the stream (e.g., a 50-meter buffer out from the stream).

Riparian Corridor: the length and width of the streamside vegetated area parallel and perpendicular to the stream (e.g., a 50-meter buffer out from the stream that extends for 1000 meters alongside the stream).

Risk: a measure of the probability that damage to life, health, property, and/or the environment will occur as a result of a given hazard.

Sedentary: staying in one place.

Sedimentation: the process by which soil particles (sediment) settle to the bottom of the stream channel. Excessive levels of sedimentation create an unstable and continually changing environment that is unsuitable for many aquatic organisms.

Stream Order: a hierarchical classification of streams. The smallest, permanently flowing stream is termed first order, and the union of two streams of order "n" creates a stream order of "n+1".

Surrogate Endpoint: a closely related endpoint to be used when data relating the assessment endpoint to human activities are not available.

Surrogate Indicator: a closely related indicator to be used when data relating the assessment endpoint to human activities are not available.

Turbidity: a cloudy condition in water due to suspended silt or organic matter.

Type I Error: Falsely concluding that there is no effect, when one is actually occurring.

FOREWORD

Risk assessment is playing an increasingly important role in determining environmental policies and decisions at the U.S. Environmental Protection Agency (EPA). EPA published *Guidelines for Ecological Risk Assessment* (U.S. EPA, 1998) to provide a broad framework that could be applied to a range of environmental problems associated with chemical, physical, and biological stressors. As ecological risk assessment evolves, it is moving beyond a focus on assessing the effects of simple chemical toxicity on single species to the cumulative impacts of multiple interacting chemical, physical, and biological stressors on populations, communities, and ecosystems. Although EPA has considerable experience in applying the ecological risk assessment paradigm in source-based approaches (such as those focused on particular chemicals), specific guidance on "place-based" approaches (e.g., watersheds and regions) is still limited.

This assessment of the Clinch and Powell Valley watershed was completed to address a specific environmental problem through application of the risk assessment approaches represented in the guidelines. Through this assessment, and other watershed scale assessments like it, the Office of Research and Development is learning how to develop new tools and approaches to support local environmental decisionmakers. An important component of these approaches is active participation by local stakeholders. The Clinch-Powell assessment provides a good example of partnering between government, environmental organizations, and others to support environmental decision making with strong science.

The Clinch and Powell Valley site was selected because the watershed contains valued and threatened ecological resources; it had previously collected stressor and effects data; it is subjected to multiple physical, chemical, and biological stressors; and a number of organizations are working to protect the ecological resources. This assessment is intended to address concerns by analyzing stressors and the resulting ecological effects and to stimulate broader public awareness and participation in decision making for reducing ecological risks. This watershed assessment report serves as an example on how to use ecological risk assessment principles in a watershed scale to improve the use of science in decision making.

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PREFACE

The National Center for Environmental Assessment, Washington, DC (NCEA–W), The Nature Conservancy, the U.S. Fish and Wildlife Service, the Tennessee Valley Authority and other organizations developed this watershed ecological risk assessment to help protect the native mussels and fish of the Clinch and Powell Valley watershed. The document has three purposes: (1) to provide information to help make more informed decisions on how to protect the valued ecological resources of the watershed, (2) to provide data and references for future research in the watershed, and (3) to demonstrate the benefits of applying ecological risk assessment at the watershed scale. The report is based on the *Guidelines for Ecological Risk Assessment* (U.S. EPA, 1998) and advice and support from NCEA, while exercising the necessary flexibility to implement the risk assessment approach at the watershed scale. To serve as an example for others seeking to increase the use of science in place-based decision making, the document includes brief descriptions of the process the workgroup followed and the major analyses performed even if analytical deliberations were not always conclusive. The literature search supporting the document was completed in May 2000.

A more concise report of the assessment's findings and methods can be found in Diamond and Serveiss (2001). A discussion of how this assessment combined ecological risk assessment with geographical information systems and multivariate analysis as tools to diagnose relationships between environmental stressors and ecological effects is presented in Diamond and Serveiss (2002). Lessons learned about applying ecological risk assessment to the watershed scale, including those acquired from this assessment, are described in Serveiss et al. (2000) and Serveiss (2002). Discussion on how ecological risk assessment principles can be applied at an even larger spatial scale (e.g., a region) can be found in Landis and Wiegers (1997) and Wiegers et al. (1998).

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The National Center for Environmental Assessment (NCEA) within U.S. Environmental Protection Agency's (EPA's) Office of Research and Development was responsible for preparing of this document. The draft report and much of the analytical work was prepared under Purchase Orders Nos. 8W-1421-NASA, OW-0399-NASA and 1W-0903-NALX with Tetra Tech, Inc., Owings Mills, MD, and reviewed through Purchase Order No. 68-C-99-238 with Versar, Inc., Springfield, VA. Some toxicity testing used for the analysis was performed under Inter-Agency Agreement #DW14937951-01-2 between EPA and the U.S. Fish and Wildlife Service.

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The authors wish to acknowledge the efforts of the following people: Suzanne Marcy of NCEA for starting the assessment, John Miller of the Office of Water for managing the group through problem formulation, Dennis Yankee of the Tennessee Valley Authority and Jeffrey White of Tetra Tech, Inc., for helping with the GIS analysis, and Forest Rich of the Tennessee Valley Authority for facilitating some of the meetings. The authors also wish to acknowledge the efforts of Terri Konoza of NCEA for managing the document production activities, Patricia von Brook of the KBM Group for editing support, and Kate Stinson and Heidi Volf of the KBM Group for word processing support. Since none of the four authors were leaders when the assessment started, we especially wish to thank those listed as contributors and other workgroup members for providing us with the necessary continuity.

1. EXECUTIVE SUMMARY

The Clinch and Powell Valley watershed assessment provides documentation to confirm suspicions of resource managers that mining, urbanization, and agricultural activities were adversely impacting native fish and mussels. Resource managers can now make more informed decisions when selecting actions to protect ecological resources. Analyses showed, for instance, that 55% of the variability in the fish community could be explained by land use, with mining and urban land uses as the most influential factors.

The document has three purposes: (1) to provide information to Federal, State, and local organizations to help them make more informed decisions on how to protect the valued ecological resources of the watersheds of the Clinch and Powell rivers, (2) to provide a repository of literature and analytical efforts for future research in the watershed, and (3) to provide an example for other watershed and regional assessors seeking to increase the use of environmental monitoring and assessment data in decision making.

Ecological risk assessment is a process for collecting, organizing, and presenting scientific information to make it more useful for decision making. The process is a unique form of ecological assessment and includes the term "risk" because it presumes that a cause and effect relationship exists and that the relationship can be expressed as a stressor-response curve. The executive summary and the report itself are organized in part according to the ecological risk assessment process, which consists of planning, problem formulation, risk analysis, risk characterization, and risk communication.

The Clinch and Powell rivers originate in the mountainous terrain of southwestern Virginia and extend into northeastern Tennessee, flowing into the upper reaches of the Tennessee River. The watershed covers 9,971 km^2 and historically has contained one of the most diverse fish and mussel assemblages in North America. Most of these populations have declined dramatically or been eliminated.

For this risk assessment, an interdisciplinary, interagency workgroup of scientists and resource managers was established. The U.S. Fish and Wildlife Service (FWS), the Tennessee Valley Authority (TVA), The Nature Conservancy (TNC), the Virginia Department of Game and Inland Fisheries, the Virginia Cave Board, the Virginia Department of Conservation and Recreation, the U.S. Environmental Protection Agency (EPA), and the U.S. Geological Survey (USGS) were represented. The workgroup was co-chaired at various times by representatives of TNC, EPA, USGS, and FWS.

The workgroup developed a management goal and selected assessment endpoints to analyze the optimal suite of data to be useful for decision making. The management goal was to:

Establish and maintain the biological integrity of the Clinch and Powell watershed surface and subsurface aquatic ecosystem.

The workgroup recognized that the valued ecological resources of concern were the diversity and abundance of native aquatic macroinvertebrates (especially mussels), fish, and cave fauna. As data on cave fauna were lacking, two assessment endpoints were selected: (1) reproduction and recruitment of threatened, endangered, or rare native freshwater mussels and (2) reproduction and recruitment of native, threatened, endangered, or rare fish species. The assessment endpoints were selected on the basis of their relevance to the management goal, their susceptibility to stressors, and their ecological importance. The workgroup acknowledged that the two assessment endpoints are linked, because most native mussels require a fish host in part of their life cycle. Because data on mussel species were limited in this assessment, data on an appropriate surrogate indicator, the fish Index of Biotic Integrity (IBI) (a series of measures of the fish species present at a sampling site that collectively describe the quality of the fish community) was used.

The workgroup agreed to focus the assessment on the unimpounded stream segment above Norris Lake, as only that portion of the watershed provides suitable habitat for the fish and mussel species of concern. The assessment analyzed data previously collected by the TVA's Clinch Powell River Action Team Survey (CPRATS) and Cumberlandian Mollusc Conservation Program (CMCP).

Conceptual models were developed by the workgroup to show the pathways between sources, stressors, and direct and indirect ecological effects. The models also helped identify and select the most important pathways for analysis, relationships between assessment endpoints and sources of uncertainty. The model was later redrawn on the basis of new information, and this helped define and prioritize subsequent analyses.

The analysis plan for this risk assessment was developed from the conceptual model and existing concerns or risk hypotheses. It was surmised that agricultural land use would correlate with various habitat measures, such as sedimentation. It was also surmised that the habitat measures would correspond to biological measures representative of the assessment endpoints. It was also believed that impacts from mining and urbanization would have some impact on habitat quality and, in turn, on biological data. Episodic spills were also thought to have impacted the valued biota but that this would be difficult to prove quantitatively because water quality data for the period shortly after spill occurrences were not available. However, qualitative data that were based on other published literature are incorporated into the conclusions. It is well established that riparian (stream-side forested) buffers help mitigate adverse impacts from human activities; however, for this mountainous region, it was unknown how wide and continuous these buffers would need to be to provide benefits. Knowing the

association between larger buffers and expected improvements to aquatic fauna helps managers decide whether and to what degree to maintain a vegetated stream buffer, because they must weigh the costs of restoring riparian buffers against the benefits provided.

Forward stepwise multiple regression analyses and/or univariate statistical analyses of data within a geographical information system (GIS) were used to test hypotheses and the strength of stressor-response relationships in order to characterize the risk to valued ecological resources (assessment endpoints). GIS maps were produced to help examine risk hypotheses.

Biological measures were used to characterize fish, mussel, and macroinvertebrate data. For fish, the fish IBI, was used. Mussel data were measured by species richness (number of different species) and abundance (the number of mussels present). Macroinvertebrate data were measured by the number of taxonomic families of *Ephemoptera* (mayfly), *Plecoptera* (stonefly), and *Trichoptera* (caddisfly) (EPT) present at a site. These three orders of macroinvertebrate are known to be sensitive to adverse water quality and are replaced by other macroinvertebrates as water quality diminishes. Several habitat-quality measures, including bottom sediment characteristics, bank stability, riparian vegetation integrity, and channel morphology, were also used to characterize habitat-related stressor exposure.

A pilot study was used to test the proposed analytical approach using a single subwatershed. Subwatershed analysis was considered a useful analytical approach because different subwatersheds in the Clinch and Powell basin had a different complement of human activities and, therefore, stressors present. Copper Creek was chosen for this pilot analysis because it was the most data-rich subwatershed and because it was a relatively simpler case in that agricultural uses were the only source of anthropogenic activity. In the pilot study, two analytical objectives central to this assessment were tested, refined, and found to be useful: (1) the appropriate spatial scale to test relationships between land-use activities or stressors and measures of effect was generally determined and (2) the fish IBI was found to be a reliable surrogate measure of effect for predicting the status of native mussel assemblages. Achieving the latter objective was especially desirable, because it was known at the outset of this study that available native mussel data were more limited than either EPT or IBI values.

Besides helping to confirm and refine the methods for performing the watershed assessment, results from the pilot study of Copper Creek provided documentation to confirm suspected beliefs. The results listed below indicate that, in this subwatershed, riparian corridors need to be protected with natural vegetation (preferably forest) and that effects of human activities can be dramatic and far-reaching downstream.

- Instream habitat quality and biological integrity were affected more by agricultural land uses very close to the stream than by agricultural land uses further away (upland),

- Impacts on native fish and mussels from agricultural land use were distinguishable for up to one mile downstream,

- Fish community integrity reflected impacts on native mussel species,

- Biological measures were found to be more strongly related to land use than to habitat measures, and

- A riparian corridor zone 200 meters across (100 m to either side of the stream) and extending 500 to 1500 meters upstream is the optimal spatial area to work with when analyzing land-use effects on fish and mussels.

The most successful analytical approaches in the Copper Creek pilot study were applied to the entire watershed. Because other parts of the watershed are subjected to stressors from the coal industry and urbanization, the riparian land cover analyses were expanded to include

- Location of different types of mining activities;

- Location of biota relative to urban/industrial areas;

- The percentage of riparian and upland land use that was forest, pasture, cropland, or urban; and

- Location of three classes of roads, including major U.S. highways, State roads, and county roads.

Several types of analyses were performed, and some were found to be more useful than others. All of the analyses are presented in the report to illustrate the efforts a workgroup may wish to undertake in performing such an assessment. A summary of major findings from the various analyses is provided below.

<u>Effects of land use on habitat quality.</u> Forty-two percent of the variability in habitat quality measures could be explained by upstream land uses within the riparian corridor at a given site. Stream sedimentation was lower where cropland was > 3% of total land use. Riparian integrity was better in areas in which pasture or herbaceous land was < 50% of the total land use. Instream cover was poor if urban use was > 20% of the surrounding area upstream. Instream cover and the degree to which the rocks in the stream were surrounded by particulate matter

(embeddedness) were affected by both the percent pasture herbaceous cover as well as the percent of urban area nearby.

Relationships between land use and biological measures of effect. Together, riparian land uses accounted for 55% of the variability in IBI scores among sites with proximity to mining as the most influential factor. Percent pasture area was directly related to the IBI, whereas proximity to mining and percent urban land were inversely related. The apparently positive effect of pasture land on the fish IBI was unexpected because of the negative relationship between pasture area and riparian integrity. This occurred because mining sites were typically near forested areas; therefore, in this analysis, higher fish IBI scores were associated with areas that had *less* forested cover and more agricultural land. The number of native mussel species was related to several land uses, including (in order of significance) percent urban area, proximity to mining, and percent cropland. However, only about half as much variation in mussel species richness could be explained by land use (26% vs. 55% for the fish IBI).

Relationships between habitat quality and biological measures of effect. Less of the variance in the IBI (29%) could be explained by available habitat quality data, as compared to land use. Embeddedness (or the inverse, clean sediment) and instream cover were most clearly related to the fish IBI, particularly if the IBI was categorized as either poor or good, based on TVA's criteria. Sites with either high substrate embeddedness scores or low instream cover scores had greater than a 90% chance of having poor fish community integrity.

Cumulative stressor index for each site. A cumulative stressor index for each site was developed on the basis of how many significant sources of stress were within 2 kilometers of the site. The four stressors were proximity to mining activities, proximity to urban areas, proximity to major transportation corridors, and percentage of cropland area in the upstream riparian zone. The fish IBI was inversely related to the cumulative number of stressors present and was consistently poor or very poor (TVA rating) at sites where all four stressors were present. Approximately 66% of the sites that had two of the four stressors present had low IBI scores, indicating poor fish community integrity at those sites, according to TVA. In nearly all of these cases (88%), the stressors present were proximity to urban areas and mining. Similar results were found for the maximum number of mussel species present at a site. Sites that had two or more stressors had greater than a 90% probability of having fewer than two mussel species present. Sites with one or no sources of stress had between 4 and 18 species, which is still far less than the number historically reported.

Riparian corridor dimension analysis. Analyses of mussel data from Tazewell County indicate that riparian land uses can have varying effects on biota, depending on landscape factors such as slope, elevation, and stream size. Results of the analyses support the riparian corridor

dimensions used in the assessment. Riparian corridor zones (5–10 km) may be required to preserve or restore mussel communities in small, high-slope streams; shorter zones (1–2 km) of riparian protection may be adequate in larger, lower-gradient streams.

Biota in Copper Creek are adversely impacted by agricultural activities; there are no mining or substantial urban activities in this subwatershed. The data for the Clinch and Powell watershed as a whole indicate that mining activities, followed by urbanization, are causing the greatest adverse impacts on fish and mussel species. Because mining was such a big factor, we investigated the adverse effects caused by different types of mines. Coal processing plants appeared to have the greatest effect on fish communities, as compared with other types of mining or other land-use activities. Thus, stressor impacts appear to be related more to water contamination than to physical habitat effects. Similar effects were documented for mussels and other invertebrates.

More stressors were observed to co-occur as we progressed upstream in this watershed, due to greater coal mining activity and associated transportation corridors and urban centers in headwater areas, particularly in the Powell River portion of the basin. Episodic chemical or coal slurry spills, although low in frequency and duration in this watershed, have undoubtedly had a significant impact on mussel and native fish species abundance and distribution. Many of these spills have also occurred in headwater areas of the watershed. Therefore, tributary and headwater populations, which were historically some of the richest faunal locations in the watershed, are most at risk from extirpation because native species migration and recruitment could be more difficult.

Several lines of evidence point to the importance of various land-use activities and riparian corridor integrity as determinants of native mussel and fish distribution in the Clinch and Powell River basin. Key factors appear to be sedimentation and other forms of habitat degradation from urban and agricultural areas as well as toxics from coal mining operations and urban areas. Riparian areas with more forested land cover and less cropland, urban, or mining activity tended to be associated with less sedimentation, more instream cover for aquatic fauna, less substrate embeddedness, and higher fish and native mussel species richness. Our results suggest that if agricultural or urban use upstream is great enough within the riparian zone, sedimentation effects and subsequent loss of habitat will ensue for some distance downstream (1–2 km).

Although riparian vegetation can reduce deleterious land-use effects on water quality, it is not clear that improvement of the riparian corridor alone in this watershed will necessarily result in recovery of native mussel and fish populations. Little or no recovery of threatened or endangered mussel or fish species has been observed in this basin despite improved water quality.

Results of the risk assessment suggest that the risk of extirpation of native species is likely to increase as more sources of potential stress co-occur. Previous analyses have indicated that none of the present mussel concentration sites (i.e., known sites containing relatively large numbers and kinds of native mussels) are located where coal mining activity is present, and only about half of the mussel sites appear to be reasonably isolated from major roads, urban areas, mines, and agricultural areas. This information suggests that native mussel populations are relatively vulnerable to likely sources of stress in this watershed and that further extinctions or extirpations are probable unless additional resource protection measures are taken.

Native fish and mussels are at high risk because of habitat fragmentation, which results in populations that are too inbred, small in size, and more susceptible to stressors. Populations are now more widely separated than they were historically, which could lead to reduced recruitment success and declining populations, especially in the presence of stressors. Therefore, it may be most useful to further protect those populations that appear vulnerable due to proximity to mining, urban areas, or transportation corridors. Protection and/or enhancement of the riparian corridor at these sites, as well as protection from toxic spills and discharges, is probably as important in terms of sustaining native species as stocking new or historically important areas. If stream habitat as well as water quality can be maintained or improved, present mussel and fish populations might be able to expand into nearby areas, thus increasing the distribution and abundance of these species.

Several uncertainties precluded our ability to describe stronger associations between causes and effects. First and foremost, although there was a lot of biological information available in the Clinch watershed to work with, it was not very often associated in time and place with relevant instream chemical or habitat measurements. Because data support the adverse impacts of spills on mussels, and to a lesser degree fish, the lack of water chemistry data, especially during spill events, posed problems when attempting to draw associations between biological condition and known or potential stressors. Second, physical habitat assessment data were fairly qualitative and relatively infrequent. Third, the macroinvertebrate measure EPT was associated with a moderate degree of uncertainty, perhaps because family-level taxonomy was used, resulting in a relatively narrow-ranging index throughout the watershed. Fourth, the potential relationship between fish IBI and mussel species richness or abundance observed in the Copper Creek subwatershed could be explained in more detail than was possible in this risk assessment. The IBI is composed of a number of metrics, one of which is native species richness. We were unable to obtain individual IBI metric values for all sites, though these data do exist. With additional effort, these data could be obtained and compared with available mussel data.

The risk assessment has helped lend further credence to what many resource managers have long conjectured were problems within the watershed, thereby providing more scientific support for taking actions to address problems. Based on the assessment findings, conservation agencies and organizations are considering riparian buffer protection; spill prevention devices along transportation corridors near streams and restriction of the type of materials transported over certain bridges; limited access of livestock to streams; better monitoring and control of mine discharges to streams; maintenance of existing natural vegetation; best management practices (BMPs) for pasture and agricultural land to reduce sediment loading; and better treatment of wastewater discharges.

During the assessment, information from several different sources was compiled and organized into a usable data set. The data set will be useful to FWS, TNC, and others as they strive to develop plans and make decisions regarding actions to further the recovery of endangered and rare species. It will also benefit other environmental agencies and organizations because they can more easily add to and use the data to further assess problems for other decision-making purposes.

The analyses also provided suggestions for future data collection to make the data more useful in decision making. For macroinvertebrates, resource managers should consider using lower-level taxonomy (genus or preferably species) and developing a suite of sensitive reliable metrics that are demonstrated to respond to human activities. In addition, as only eight sites in the entire watershed had mussel and IBI or EPT data, taking samples of all fauna at each site (along with more robust habitat assessment measures) would reduce uncertainty in observed biological effects.

2. BACKGROUND

2.1. WATERSHED DESCRIPTION

The Clinch and Powell rivers originate in mountainous terrain of southwestern Virginia and extend into northeastern Tennessee, flowing into the upper reaches of the Tennessee River (Figure 2-1). They collectively cover 9,971 km^2. The Clinch and Powell watershed historically has contained one of the most diverse fish and mussel assemblages in North America (Neves, 1991), yet most of these populations have declined dramatically or been eliminated (Neves et al., 1985). The mainstem Tennessee River and many of its tributaries have been dammed, resulting in the loss of habitat for many fish and mussel species (Yeager, 1994). However, the upper regions of the Clinch and Powell rivers represent some of the last free-flowing sections of the expansive Tennessee River system. Currently, the Clinch and Powell river basin supports more threatened and endangered aquatic species than almost any other basin in North America (Stein et al., 2000). Despite the implementation of recovery plans for most of the federally protected species in this basin, there is evidence that these species are either declining or becoming extinct at an alarming rate due to impacts from mining, agriculture, urbanization, and other stressors (Jones et al., 2000).

The upper regions of the Clinch and Powell rivers drain approximately 7,542 and 2,429 km^2 (2,912 and 938 sq. mi.), respectively, and vary in elevation between 300 and 750 m. Both rivers flow southwesterly through parallel valleys and are contained within the Cumberland (Appalachian) Plateau and the Valley and Ridge physiographic provinces (UCS, 1992) (Figure 2-1). These two subwatersheds are characterized by steep slopes and poor riparian forest cover, suggesting relatively high vulnerability of aquatic species to anthropogenic stressors. The climate of the Clinch and Powell watershed is moderate, with an average temperature of 12.0 ˚C. Precipitation varies across the region, from 96.5 to 127 cm. Wide variability in both precipitation and soil types at the local level is common throughout the region and leads to a high degree of plant diversity. The watershed is composed largely of forest and agricultural land, although there are several small urban/industrial areas scattered throughout the basin (see section 2.2).

The Clinch River begins in Tazewell County, VA, and flows for approximately 321.9 km (200 miles) before reaching Norris Lake. The majority of the Clinch drains the Valley and Ridge province, although several of the river's western tributaries drain the Cumberland Plateau. The geology of the Clinch River basin is characterized by large expanses of limestone and dolomite, resulting in large areas of karst topography (a limestone region characterized by caves and underground channels). Daily average flow data—the average volume of water that flows past a

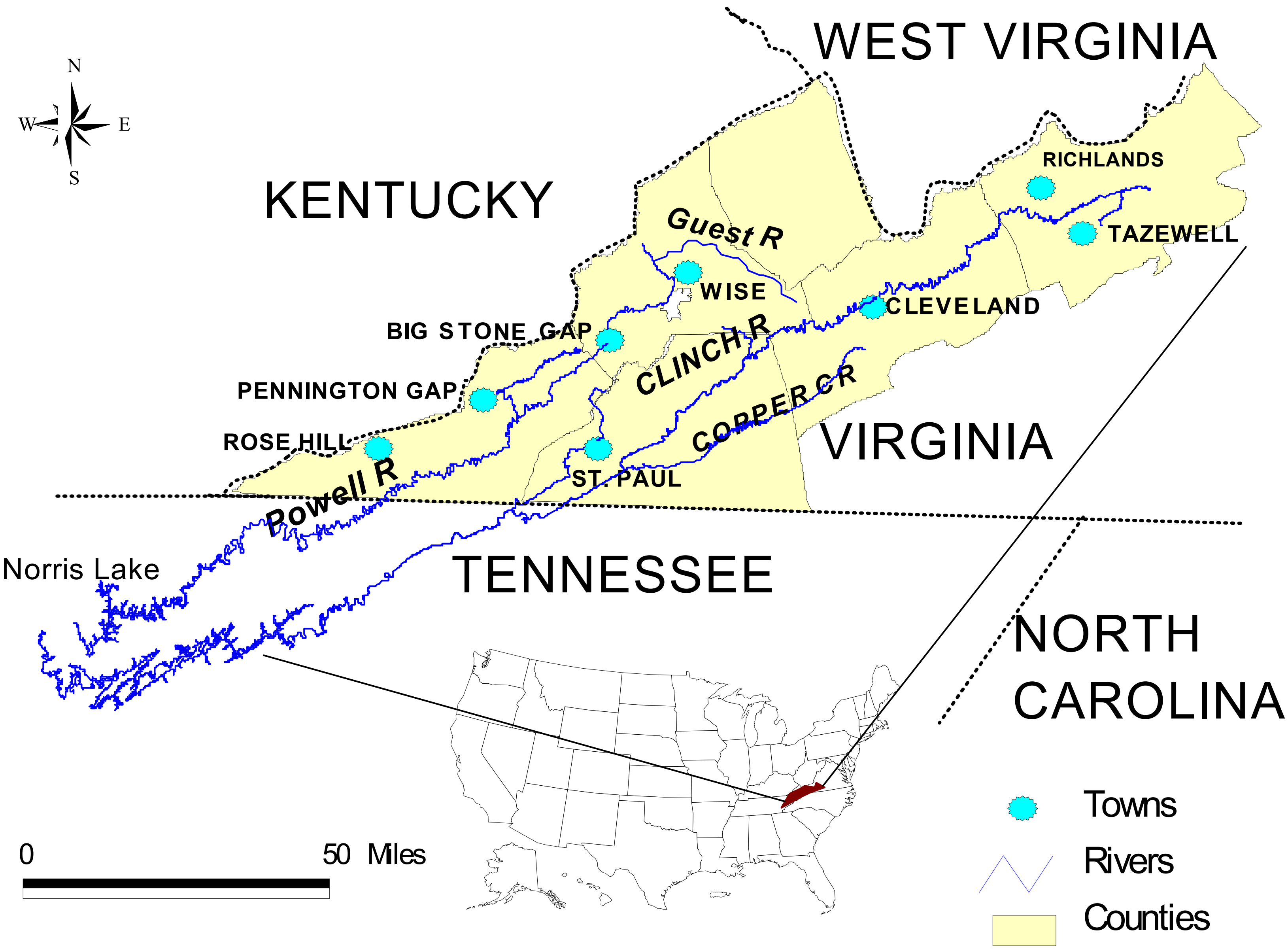


Figure 2-1. Watershed boundaries, major cities, and subwatersheds examined in this risk assessment.

given point for 1 second—show that the Clinch River in Virginia ranges from 5.38 m^3/sec at Richlands, near the headwaters, to 45.1 m^3/sec at Clinchport. The 7-consecutive–day low flow on average in a 10-year period (7Q10) is 0.44 and 2.82 m^3/s for Richlands and Clinchport, respectively.

The Powell River, a tributary to the Clinch River, begins in Wise County, VA, flows approximately 193.1 km (120 miles), and enters Norris Lake. The headwaters of the Powell, including the mainstem and tributaries, primarily drain the Cumberland Plateau. As the Powell leaves the Cumberland Plateau, it enters the Valley and Ridge province, which is characterized by extensive parallel ridges with valleys of varying size. Extensive subsurface drainage is common, with broad areas of karst dotted with caves, sinkholes, and sinking streams. Major tributaries to the Powell include the South and North Fork Powell rivers. Flow in the Powell River in Virginia ranges from 3.68 m^3/sec on the North Fork Powell near Pennington Gap to 15.21 m^3/sec at Jonesville, with 7Q10 values of 0.03 and 0.69 m^3/sec, respectively.

Because of the mountainous terrain, the watershed maintains a rural character, with a population of approximately 170,000 (1990 census) in Lee, Scott, Wise, Russell, and Tazewell Counties making up the Virginia portion of the watershed. The urban areas in the watershed include Wise, Norton, Pennington Gap, Rose Hill, and Big Stone Gap in the Powell River subwatershed and Tazewell, Rocklands, Cleveland, and St. Paul in the Clinch River subwatershed.

Untouched by either glaciation or rising seas in recent geologic time, and isolated from other nearby river systems, the assemblage of fish and freshwater mussel species in the upper Clinch and Powell rivers is among the most diverse in North America (Ortmann, 1918; Ahlstedt, 1991). The decline of these native species is accentuated by the fact that native mussels evolved to depend on fish. Unionids (mussels) are sedentary filter-feeding macroinvertebrates that burrow into a gravel/cobble substrate and remove unicellular algae, zooplankton, detritus, and silt from the water column (Neves, 1991). They have a unique life cycle that includes an obligate parasitic larval stage, or glochidia, that must attach onto the fins, epidermis, or gills of a suitable host fish (Bogan and Parmalee, 1983) (Figure 2-2). The host fish is apparently unaffected by glochidia parasitization; however, some mussel species can parasitize only certain fish species (Zale and Neves, 1982a, b). Large numbers of glochidia are released: 100,000–3.5 million either in spring or midsummer, corresponding not only with the migration and spawning activities of many resident fish species, but also with relatively low stream flow and potentially high concentrations of toxic chemicals (Zale and Neves, 1982a, b; Kitchel, 1985). The relatively low occurrence of glochidia on host fish indicates that most do not reach this point in the life cycle.

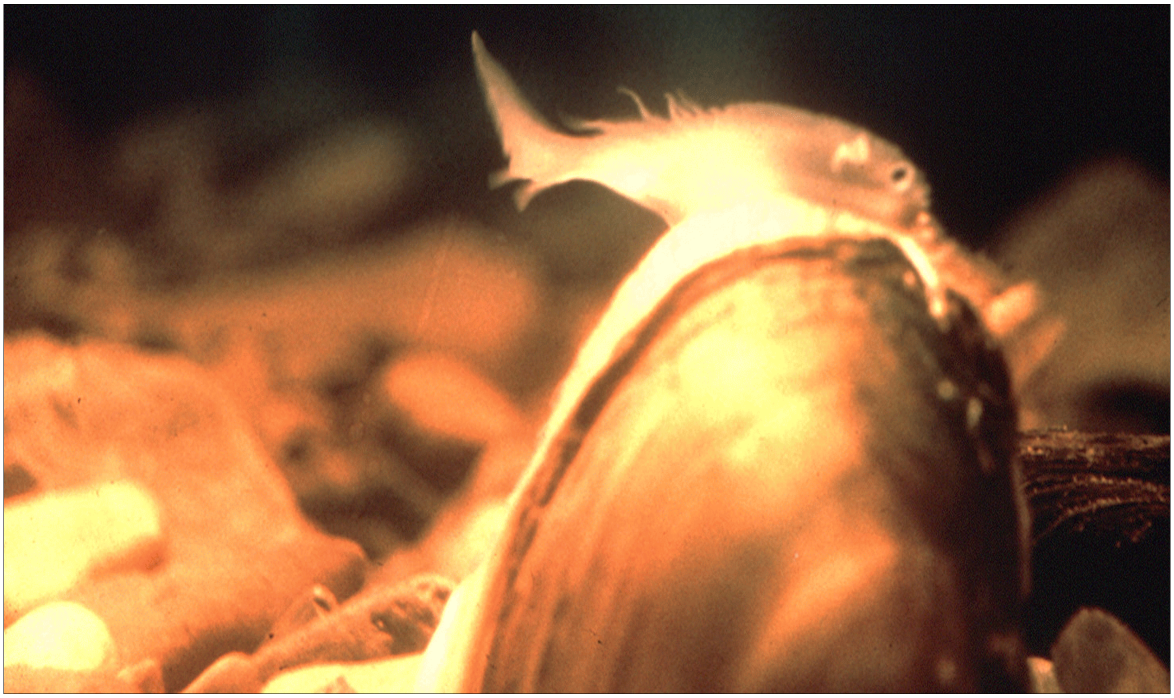


Figure 2-2. Many endemic mussel species in the Clinch and Powell watershed have evolved sophisticated anatomical structures to attract the appropriate fish host. The correct fish host is needed to complete the mussel's life cycle and enhance species dispersal.

The codependence of native mussels on native fish species is believed to have evolved over millions of years, suggesting that glochidial infestation success rate, though small, is adequate to ensure the dispersal of mussel populations. Following the 1- to 3-week parasitic phase, the glochidia drop to the substrate and begin their sedentary free-living phase. Owing to the fact that they are sedentary and that they have a complex life cycle, unionid mussels cannot readily migrate or recolonize new stream areas, except during the larval parasitic stage. The potential results are geographically isolated populations, genetic inbreeding, and reduced adaptive potential (Stansbery et al., 1986; Ahlstedt, 1991). Clearly, the survival of unionids is dependent, in part, on the reproductive success and distributional range of the appropriate host fish species (Zale and Neves, 1982b; Young and Williams, 1983; Neves et al., 1985; Watters, 1997).

Previous assessments of Virginia's aquatic biota indicate that the Clinch and Powell watershed supports more imperiled mussel and fish species than most streams in North America (Neves, 1991; Jenkins and Burkhead, 1994; Stein et al., 2000). Recent assessments have reported continued declines and possibly extirpations of native species in several areas of the watershed (Ahlstedt, 1999; Jones et al., 2000).

Although recovery plans have been developed for most federally protected species in the Clinch and Powell rivers, evidence of recovery is lacking (Sheehan et al., 1989; Jones et al., 2000). Fish and mussel surveys by biologists in Virginia and Tennessee indicate that most rare species in this region continue to decline (Angermeier and Smogor, 1993; Ahlstedt, 1999). This degree of loss is unprecedented among other wide-ranging faunal groups in North America (Neves, 1991). Thus, the Clinch and Powell watershed, one of the few remaining refugia for these fauna, has national significance. Reversing the decline or loss of these rich faunal groups is a test of our commitment to preserving biodiversity on a national scale.

Given (1) its importance as a center of aquatic biodiversity; (2) the profound, diverse, and yet unquantified effects of human activities; (3) the relatively large amount of existing biological and land use data; and (4) the ongoing efforts of TNC, FWS, and other organizations, the Clinch and Powell watershed was selected by EPA as one of several national watershed ecological risk assessment case studies.

The intent of this project was to (1) collect, organize, analyze, and present available ecological information to assist resource managers in the Clinch and Powell watershed to improve their decision making; (2) increase the likelihood that available (and often limited) environmental monitoring and assessment data will be used appropriately in decision making; and (3) serve as an example for others seeking to integrate ecological risk assessment with a watershed or place-based approach.

2.2. LAND USES IN THE WATERSHED

The economy is driven primarily by coal mining and agriculture. More than 40% of Virginia's coal production lies within the five counties in the basin, where the Cumberland Plateau is composed of Pennsylvanian sandstone and shale; the remaining 60% is in adjacent Buchanan and Dickenson counties. Coal production increased from 1980 to 1988. The region's coal supply is estimated to last for another 25 to 50 years. There are 287 active point-source discharges from coal processing plants and mine sites, and only a few potentially toxic chemical contaminants in these discharges are regulated. The upper Clinch River in the vicinity of Swords Creek, the Guest River, and the upper Powell River upstream from Pennington Gap have been heavily impacted by sediment, coal fines (fine particulate coal and refuse rock material), and acidic runoff from mining activities.

Most of the watershed was intensively logged to clear land for agricultural production in the late 18th and early 19th centuries. Another logging boom flourished in the late 19th century into the early 20th century, spurred by national industrial growth as well as salvage harvest of the American chestnut, which was killed by the disease caused by the fungus *Endothia parasitica*. The logging and forest industry in general declined through the 20th century,

although there has been some production of mine timber supports for the mining industry. Quality hardwood, timber, and pulp were still exported from the region, but the 1980s and 1990s saw a resurgence of the forest industry. In the mid-1980s and early 1990s, an oriented strand board plant was active in Dungannon, VA. Several large forest industries have recently been established in Dickenson County, VA, and in nearby West Virginia and Tennessee. Many former miners or support workers to the mining industry have entered the forests of southwest Virginia to provide logs to these new forest industries. Although growth of this industry is helping to meet the economic challenges of the decrease in the mining sector, logging in some areas, without proper use of best management practices (BMPs), can pose a threat to sensitive aquatic resources.

Agriculture is the other chief economic activity in the Clinch and Powell watershed, accounting for approximately one-third of its land use. Beef cattle and Burley tobacco are the primary agricultural products (UCS, 1992). Topographic constraints limit most of these agricultural activities to the floodplain, where livestock and row crop production are most feasible and productive. Pesticide runoff, runoff from overgrazed pastures on steep slopes, animal waste from feedlots, and livestock access to streams and riparian corridors threaten water quality and ecosystem integrity in the watershed. Aquatic and subterranean ecosystems are especially vulnerable as a result of increased sediment loading, nutrient enrichment, and pathogens.

Although the region is currently exhibiting slow economic growth, urban development is

Figure 2-3. The swiftly flowing waters of the Clinch River provide a source of recreation for outdoor enthusiasts.

planned in either karst terrain or the floodplain, both of which are sensitive to alteration. For example, an airport is planned in the Central Lee County karst, perhaps Virginia's most biologically significant karst area. Despite existing stressors, the region's environmental resources provide recreational opportunities (Figure 2-3).

3. PLANNING AND PROBLEM FORMULATION

In this watershed, as in any other, the optimal suite of data is not available, and yet managers must make decisions on the basis of existing information. The purpose of risk assessment is to analyze available scientific information and present it in a manner that enables more informed decision making on how to protect the unique biological resources of the Clinch and Powell watershed. Before starting the risk assessment, some planning is required. During planning, the managers define the management goals for the watershed (U.S. EPA, 1998). Then, managers, in consultation with the scientists performing the assessment, reach agreement on the purpose, scope, and complexity of the risk assessment. The risk assessment begins with problem formulation, during which risk hypotheses, conceptual models, and a plan for risk analyses are developed (U.S. EPA, 1998) (Figure 3-1). Next, the analysis phase evaluates the exposure of valued ecological resources to stressors and the relationships between stressor levels and ecological effects. During risk characterization, exposure and effects data are integrated to describe risks and draw conclusions (U.S. EPA, 1998). In this risk assessment, the workgroup met periodically to share interim findings and refocus remaining analyses. Thus, risk was characterized and presented several times, and progressively in more detail each time. Furthermore, the workgroup directly participated in guiding risk analyses as well as risk characterization in this assessment.

3.1. MANAGEMENT GOALS AND RISK ASSESSMENT OBJECTIVES

Federal, State, and local managers have been working with scientists to study the extent of water quality alteration in the watershed. The Tennessee Division of Water Pollution Control conducts frequent water chemistry and benthic community surveys in both the Clinch and Powell rivers inside Tennessee near Norris Lake. The Virginia Department of Game and Fisheries, the Virginia Cooperative Fish and Wildlife Research Unit, the Tennessee Wildlife Resources Agency, and TNC are also conducting studies of water chemistry, mussels, fish, and riparian vegetation in the Clinch and Powell watershed. The Virginia agencies listed above and FWS are responsible for protecting endangered species in the area. These agencies also have been working to educate stakeholders about the unique aquatic and other natural resources and about efforts to protect them. In addition, TNC has established the Clinch Valley Program to conserve diversity in the Clinch and Powell and Holston River watersheds while meeting human and economic needs.

A tremendous amount of information has been collected in this watershed over many years, but much of it has not been analyzed. Monitoring data have been collected by TNC, TVA, FWS , USGS, the Virginia Department of Game and Inland Fisheries, and the Virginia

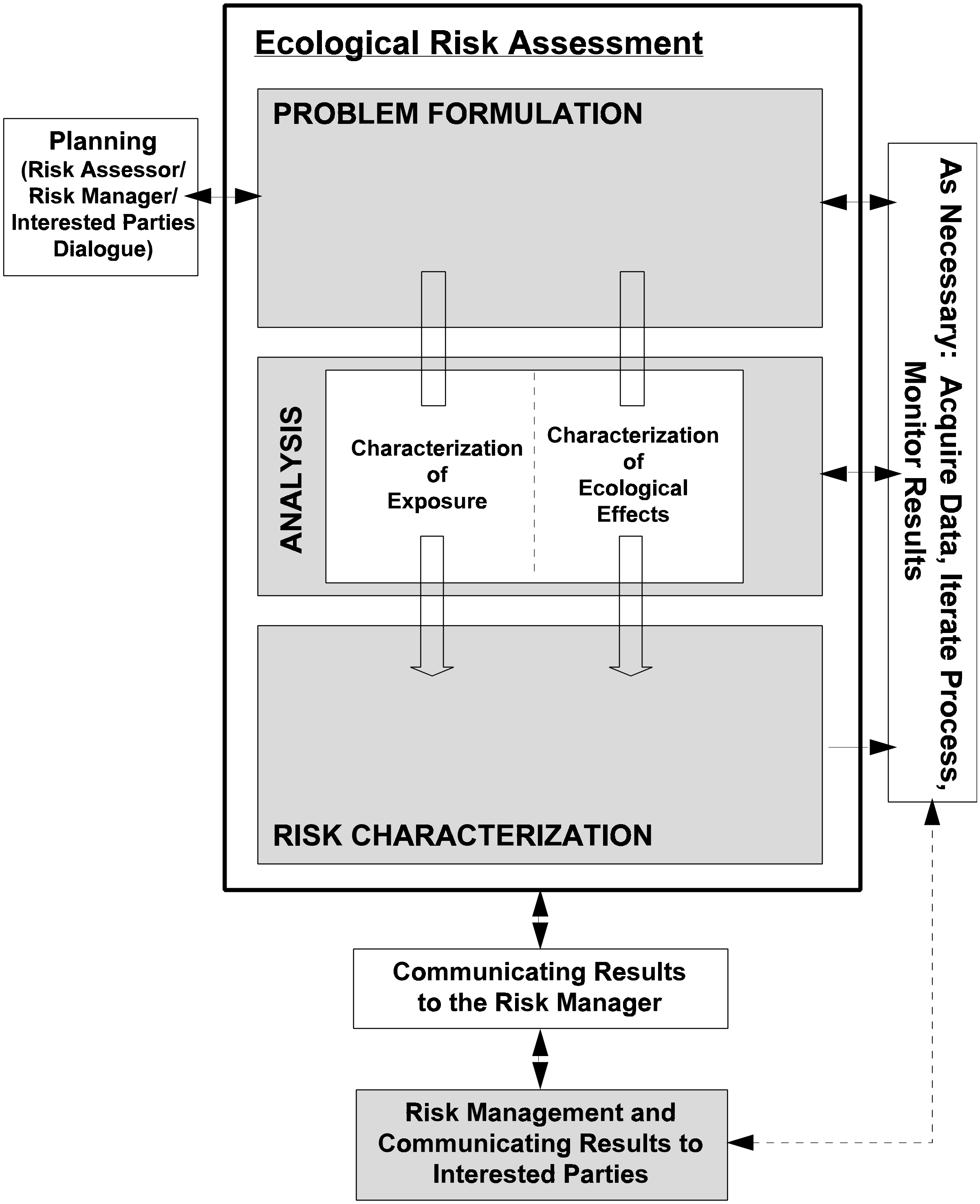


Figure 3-1. Framework for ecological risk assessment.

Source: U.S. EPA, 1998

Department of Conservation and Recreation. Although several hypotheses have been advanced to explain the decline in native mussel and fish species in other watersheds (Watters, 1996), definitive answers have been lacking. There were suspicions that many of the problems were due to agricultural, mining, and urban influences as well as episodic spills. A number of studies had been performed by scientists for various purposes, but no one had performed a compilation and analysis of the existing data using sophisticated tools such as GISs and mulitvariate analysis. Furthermore, no effort had ever been undertaken to bring many of the resource managers in the area together to help determine which analyses of pre-existing data would be most useful to improve decision making. Resource managers in the Clinch and Powell river basin recognized that a comprehensive examination of the available data was needed to evaluate the relative effects of different human activities on native mussels and fish.

The workgroup charged with designing the risk assessment for the Clinch Valley was convened at a meeting in Dungannon, VA, in 1993. The Clinch and Powell watershed ecological risk assessment workgroup has been co-chaired since 1993 by members from TNC, FWS, USGS, and EPA. (See Acknowledgments for list of workgroup members and co-chairs and Appendix A for a list of stakeholders.) At the 1993 meeting, workgroup members and other stakeholders characterized the ecological resources present in the watershed and potential problems affecting those resources. The workgroup then developed an initial description of the risks from an earlier draft document and decisions were made regarding the scope of the assessment.

TNC conducted a random telephone poll to assess the level of awareness and concerns of people in southwest Virginia. The results indicated an interest in conserving the water resources of the region and a strong sense of pride in the natural beauty of the area. Currently, the Upper Tennessee River Roundtable holds public meetings to gather information to help develop a strategic plan for the watershed. These meetings reaffirm the concerns previously voiced by stakeholders in the 1993 meeting and the TNC survey.

Figure 2-1 shows the hydrologic drainage of the Upper Tennessee River Basin and the watershed boundary for the upper Clinch and Powell rivers. Because there is a significant reduction in aquatic species diversity caused by impoundments downstream of Norris Lake (Masnik, 1974; Ahlstedt, 1984; Angermeier and Smogor, 1993), the free-flowing portion of the watershed was recognized as the best remaining habitat for native mussels and fish in this region and, therefore, the area most in need of better information and protection. Consequently, the workgroup decided to assess only the segment of the watershed upstream of Norris Lake. An overall management goal for the assessment was developed by the workgroup so that the results would be relevant to regulatory requirements and public concerns. The assessment was also

designed to ensure that assumptions, methods, and conclusions would be scientifically valid and documented. The broad management goal was to:

> *Establish and maintain the biological integrity of the Clinch and Powell watershed surface and subsurface aquatic ecosystems.*

This goal reflected the intent to establish sustainable native populations of flora and fauna for the riverine, riparian, and karst ecological communities. The workgroup determined that if biological integrity could be maintained, then water quality—the chief public concern—would also be protected. Workgroup members identified the watershed's three outstanding ecological resources:

1. The diversity and abundance of threatened, endangered, or rare native freshwater mussels.

2. The diversity and abundance of native, threatened, endangered, or rare fish species.

3. The diversity and abundance of cave fauna.

To attain the goal of biological integrity, this ecological risk assessment addressed the first two of these resources. The potential risks to terrestrial and aquatic communities in caves were not evaluated because little information on the distribution and abundance of these fauna were available. Section 3.1.2 discusses how stressors are believed to impact the abundance, diversity, and age class structure of cave fauna.

The workgroup agreed to consider implementing several management objectives to maintain or restore the threatened, endangered, or rare native freshwater mussels and fish in the Clinch and Powell watershed, pending the results of this assessment. Some management actions under consideration prior to this risk assessment were:

- Implementing BMPs, such as minimum till and treatment of feedlot waste, to reduce nonpoint-source pollution.

- Containing and treating runoff from mining activities to reduce pollutant load and sedimentation.

- Installing or improving sewage treatment facilities to reduce inputs of pollutants and nutrients.

3.1.1. Assessment Endpoints

In a watershed ecological risk assessment, the broad management goal often needs to be explicitly defined so that it provides a clear focus for the assessment. An important part of problem formulation involves the selection of ecologically based assessment endpoints that provide a link between measurable endpoints and the steps necessary to achieve the management goal (U.S. EPA, 1998). Specifically, risk assessors need to determine the ecological resources of concern in the watershed; in this case protecting threatened, endangered, and rare native mussel and fish species.

For each of the ecological resources of concern, assessors need to define a specific characteristic of interest (e.g., mussel species abundance and diversity). The combination of valued resource and ecologically relevant characteristic is called an assessment endpoint (U.S. EPA, 1998). Assessment endpoints are selected on the basis of their relevance to management objectives, susceptibility to stressors of concern, and ecological importance.

Even though this assessment focuses on fish and mussels, it is termed a watershed assessment because activities within, and impacts from, the entire watershed are considered in relation to the assessment endpoints. Furthermore, a better understanding of and further protection of these native species are likely to confer protection on many other plant and animal species in the watershed.

If data relating the assessment endpoint to human activities are not available, a surrogate indicator called a “measure of effect” is used. In this assessment, data for the assessment endpoint of mussel species diversity and abundance were limited. Therefore, data for an appropriate surrogate measure, such as mussel species richness or the fish IBI was used. By clearly defining the ultimate focus of the assessment (e.g., mussel species diversity and abundance), the uncertainties in the assessment can be better described (e.g., extrapolating between fish community integrity and mussel species richness).

3.1.1.1. *Assessment Endpoint 1*

Diversity and abundance of threatened, endangered, or rare native freshwater mussels.

3.1.1.1.1. ***Importance of endpoint.*** The Clinch and Powell watershed supports more of Virginia’s imperiled mussel species than any other basin in Virginia and most places in North America (Ahlstedt, 1991) (Appendix B, Table B-1). Therefore, protection of threatened and

endangered mussel species' and their habitats has high ecological and societal value. Diversity and abundance are the specified attributes because they provide an indication of a specie's ability to maintain viable populations over time in a given region. These attributes support maintaining self-sustaining native populations and the goal of maintaining biological diversity.

Remnants of the unique mussel assemblage exist as fragmented populations and presently occur only in a few streams in North America, including the Clinch and Powell watershed. These drainages have the greatest number of federally listed endangered aquatic species (18) and also one of the largest concentrations of endemic species (19) in the United States (Stein et al., 2000) (Appendix B, Table B-1).

3.1.1.1.2. *Risks to endpoint.* Threatened and endangered mussel species are susceptible to a range of anthropogenic disturbances as well as natural perturbations, including sedimentation, toxic chemicals, prolonged drought, low stream current velocity, and loss of riparian corridor integrity. Many of these species are particularly sensitive to these stressors during the glochidia or larval stage. Threatened and endangered mussel species are excellent indicators of benthic macroinvertebrate habitat quality and stream water quality in general (Bogan and Parmalee, 1983; Goudreau et al., 1993; Kitchel, 1995; Warren et al., 1995).

Mussels are susceptible to any land use or natural phenomenon that (1) ultimately reduces host fish survival and reproduction, (2) degrades surface-water quality, (3) reduces or eliminates usable benthic habitat, or (4) interferes with or undermines their normal filter-feeding process. Thus, mussels are at risk from a variety of human activities in the watershed, including poor agricultural practices; urban runoff; wastewater discharges; runoff from mining, poor forestry practices, roads, and other transportation corridors; and possibly competition from the introduction of exotic species such as the zebra mussel (*Dreissena polymorpha*) and Asiatic clam (*Corbicula fluminea*).

Because available data on mussels were limited primarily to mussel species richness, this measure was used as a surrogate for the assessment endpoint of diversity and abundance of threatened, endangered, or rare native freshwater mussels. The direct relationship between presence of a particular species and mussel assemblage characteristics such as diversity and abundance is plausible because many studies have demonstrated direct correlations between species richness and diversity or abundance of mussels (Dennis, 1985; Ahlstedt, 1991; Ahlstedt and Tuberville, 1997).

3.1.1.2. *Assessment Endpoint 2*

Diversity and abundance of threatened, endangered, or rare native fish species.

3.1.1.2.1. ***Importance of endpoint.*** The Southeast has the highest diversity of freshwater fishes in the United States (Etnier and Starnes, 1994). These obligate riverine fishes historically have existed in relatively stable environments (Jenkins and Burkhead, 1994), but this has changed rapidly over the past century. Some species are not able to withstand the physical and chemical alterations to their habitats that have occurred because of human activities in the watershed (Yeager, 1994). As a result, local extirpations and extinctions have taken place. About 30% of the federally listed endangered fish species and 40% of the species that are proposed candidates for listing are located in the Southeast, indicating downward trends in the quality of southeastern aquatic habitats. The free-flowing portions of the Clinch and Powell rivers upstream of Norris Dam are major refugia for many fish species endemic to the Tennessee River drainage. Of the 85 fish species reported from these systems, about one-third are federally listed as endangered or threatened, are candidates for listing, or are listed for protection by Tennessee or Virginia (Etnier and Starnes, 1994) (Appendix B, Table B-2). Because there are so many endangered, threatened, or protected species, the fish assessment endpoint is of high ecological and societal value.

Within the past century, the entire Tennessee River proper and many of its tributaries have been physically altered by impoundments, resulting in destruction and fragmentation of these rich riverine communities (TVA, 1970; Freeman, 1987; Angermeier and Smogor, 1993; O'Bara et al., 1994). Because many of the rare fish species in this watershed are insectivorous (Jenkins and Burkhead, 1994) (Figure 3-2), protection of native fish species habitat protects many of the invertebrates as well. Many of these rare fishes are primarily benthic or have specific aquatic habitat requirements for spawning and/or feeding, and most are relatively short-lived (Masnik, 1974; Etnier and Starnes, 1994). The assessment endpoint attributes of diversity and abundance are specified because the goal of self-sustaining populations of native fish species identified in this risk assessment will be evidenced by diverse, abundant populations within the watershed. Similarly to mussels, native fish species are also extremely sensitive to the negative impacts brought about by geographic isolation and immediate loss of habitats due to impoundment. Small, isolated populations of fish not only suffer from lack of gene flow but are also highly susceptible to localized extirpations from catastrophic events such as toxic chemical spills, prolonged drought and low stream flow, and high water temperatures.

The degree of native fish species recruitment is related to several habitat and water quality features that are important to the survival and reproduction of many fish and invertebrate species in the Clinch and Powell watershed. Furthermore, specific fish species serve as hosts for the obligate parasitic glochidia stage of the native mussel species, as mentioned previously. Thus, recruitment of native fish species is an important factor in the recruitment of mussel species.

Figure 3-2. The Clinch and Powell watershed harbors several endemic fish species, particularly insectivorous darters, many of which are now rare and/or threatened and endangered.

3.1.1.2.2. ***Risks to endpoint.*** Fish reproduction and recruitment are especially susceptible to sedimentation, turbidity, and exposure to toxic chemicals, each of which can result in local extirpations, leaving disjunct populations that are even more susceptible to extinction. Habitat alteration, either through riparian corridor destruction, hydrologic modification, or livestock watering, is also an important stressor for fish recruitment. There are several sources of toxic chemicals (i.e., pesticides, herbicides, metals, oils, and greases), including runoff from urban areas, row-crop agriculture, mining, transportation corridors, and silviculture areas or from atmospheric deposition. Toxic chemicals affect potential invertebrate prey as well as the fish themselves, which could result in either reduced food for fish consumption or biomagnification of certain pollutants through the food web.

Sedimentation is believed to be a potentially important stressor to native fish populations in this watershed because it reduces suitable spawning sites and, thereby, fish recruitment. It originates from a number of sources, including livestock watering; soil erosion from urban, mining, and agricultural runoff; riparian corridor modification; and poor silviculture practices. It

also has indirect effects on fish by changing the type and abundance of food items that may be available.

Direct fish habitat alteration is possible if the riparian corridor is eliminated or greatly reduced. Stream bank cover and clean benthic gravel and rubble for spawning and cover are often important habitat features for many native fish species in the Clinch and Powell watershed (Jenkins and Burkhead, 1994). These habitat features are jeopardized if the riparian corridor is degraded. Furthermore, these same features are necessary for the survival and reproduction of many of the invertebrate prey used by native fish in this watershed.

3.1.2. Impacts on Abundance, Diversity, and Age Class Structure of Cave Fauna

Although not analyzed in this risk assessment, the impacts of human activities on the abundance, diversity, and age class structure of cave fauna were qualitatively considered. Several activities in the watershed could directly or indirectly affect subsurface water quality or cave fauna habitats. Toxic chemicals originating from agricultural runoff, mining activities, or trash disposal can enter sinkholes or caves, where they can then be potentially transported over a broad subsurface area and affect multiple caves. Excess nutrients and pathogens from livestock grazing in highlands or from agriculture could also conceivably reach the subsurface system, resulting in excessive bacterial growth, anoxia, and perhaps increased disease rate of cave fauna. Degradation of water quality due to toxic chemicals, excessive nutrients, or pathogens is expected to have direct effects on cave fauna survival and reproduction and, therefore, their abundance, diversity, and age structure.

A second type of stress on cave fauna is habitat modification due to either drastic changes in subsurface water flow or increased sedimentation. Sedimentation due to mining and poor agricultural or silviculture activities is believed to be a potentially important stressor to cave fauna because it could lead to anoxia and habitat destruction. In addition, use of sinkholes for soil and debris disposal can cause back-flooding and siltation of ground water. The presence and effects of sedimentation in relation to cave fauna abundance and diversity are poorly documented at this time.

Effects of other potential activities in this watershed, such as transportation corridors, failed septic systems, or leaking sewers, are also poorly documented for this watershed but may be inferred from land use analysis of cave fauna data (see Analysis Plan). Hydrologic modification could conceivably alter cave and subsurface water quality, depending on the location of the activity in the watershed. Also, many of these activities could result in deleterious changes in subsurface flow or sedimentation for aquatic cave fauna.

3.2. SOURCES AND STRESSORS CONSIDERED IN THE CLINCH AND POWELL WATERSHED ASSESSMENT

Table 3-1 summarizes the various stressors, and sources of those stressors, initially considered by the workgroup. The following section discusses each potential source, the stressors that might originate from the source, and general statements about the relative importance of each source or stressor to the watershed, based on resource managers' many years of experience.

Table 3-1. Stressors and sources identified in the Clinch and Powell watershed

Stressor	Sources	
Degraded Water Quality		
Toxic chemicals	Catastrophic spills Urbanization Point-source discharges Atmospheric deposition	Agriculture Coal mining Transportation
Pathogens	Urbanization	Agriculture
Nutrients	Urbanization Atmospheric deposition	Agriculture
Physical Habitat Alteration		
Sedimentation	Coal mining Hydrologic changes Transportation	Agriculture Urbanization
Riparian modification	Agriculture Hydrologic changes	Urbanization
Instream destruction	Agriculture Hydrologic changes	Urbanization
Biotic Interactions		
Exotic species introductions	Accidental (Asiatic clam, zebra mussel) Recreational (brown trout, rainbow trout)	
Overexploitation	Other biota Over harvesting	Poaching

3.2.1. Active Coal Mining and Processing

Stressors: Toxic chemicals, sedimentation

Coal mining is restricted to the western region of the watershed, along the Cumberland Plateau. Areas known to be most heavily impacted by coal activities include the upper Clinch River in the vicinity of Swords Creek and the Guest River and the upper Powell River upstream from Pennington Gap. Although this region makes up less than 20% of the total watershed area,

the impacts of mining are evident throughout the watershed. Both nonpoint and point-source pollution impacts occur from mining.

Point-source discharges from active mines and processing plants are potential threats to the riverine ecosystem. Hydraulic fluid releases associated with mining activities have caused known fish kills (BMI, 1990). There are 287 active coal mining point-source discharges in the Clinch and Powell watershed. Discharges from coal processing plants and mine sites are currently regulated by the U.S. Bureau of Mines, and discharges are monitored for pH, total iron, total manganese, and total residue. However, a wide range of potentially toxic compounds, such as hydraulic fluids, frothing agents, modifying reagents, pH regulators, dispersing agents, flocculants, and media separators that are used in mining and coal processing are currently unregulated and not monitored, and they may be discharged to the rivers (BMI, 1990; Cherry et al., 1995). Furthermore, the compliance of these discharges is not known with certainty. Enforcement of discharge compliance is also unclear, as both the Bureau of Mines and the Virginia Department of Environmental Quality (VDEQ) may not coordinate activities adequately.

Sediment runoff and coal fines from haul roads, active mining sites, and abandoned mine lands lead to sedimentation of surface waters, which can inundate mussels in the substrate with fine sediments that may also be toxic (Sheehan et al., 1989). The Powell River, particularly above Pennington Gap, VA, was so adversely impacted from coal mining operations that it was dredged to remove contaminants. On Christmas Day of 1972, the Powell River ran black from coal fines in the water column. Coal fines, which may be transported downstream during scouring from high water flows, are a major component of the substratum in some parts of the upper Powell River. This form of sedimentation has had deleterious effects on benthic organisms, particularly freshwater mussels, which are sessile and do not recover as easily as mobile organisms such as fish and insects. Sedimentation from continued mining operations is still a significant ecological stress.

Finally, coal mining activities have led to several catastrophic spills within the past 21 years at least. Accidental releases from coal slurry impoundments and toxic chemicals from mine sumps have been recorded (BMI, 1998; Hylton, 1998) (see Catastrophic Spills, below). Effects of active coal mines and processing plants were quantitatively analyzed in this risk assessment.

3.2.2 Abandoned Mine Lands

Stressors: Sedimentation, toxic chemicals

Acidic soils exposed during mining activities cause acid mine runoff that finds its way into the river and reduces the pH of the water. If one assumes that active mining point sources

are being adequately controlled through the Federal Clean Water Act, then many of the observed impacts are coming from abandoned mine lands. More than 45,000 acres of disturbed mine lands occur within the watershed, of which 9,200 acres are abandoned mine lands developed and then abandoned before Federal controls. The projected cost to reclaim the abandoned mine lands is more than $100 million, and little or no information is available on the water quality impacts of these lands.

Extensive development of the coalfields of southwestern Virginia before the 1977 Federal Surface Mining Law's reclamation requirements has resulted in significant watershed and stream impacts from both acid mine drainage (including low pH and exceedingly high metal concentrations such as iron, manganese, and aluminum) and embedded stream conditions (from sedimentation due to barren or semibarren land condition and slope instability). In specific subwatersheds of the Powell River, metal concentrations have surpassed acute and chronic toxicity thresholds by orders of magnitude, threatening aquatic life, livestock, and humans. In one heavily impacted watershed, Ely Creek, pHs ranging from 2.5 to 2.9 resulted in measures of benthic abundance and diversity of zero and a total loss of fish. As abandoned mine lands are incompletely identified in this watershed, we qualitatively examined this source of stress in this risk assessment.

3.2.3 Urbanization

Stressors: Toxic chemicals, pathogens/nutrient enrichment, sedimentation, riparian zone modification

Historically, southwest Virginia and northeast Tennessee have suffered economically because of their geographic remoteness, rugged terrain, inadequate transportation, and poor education. Consequently, efforts are underway to encourage industrial growth in the region, as evidenced by the Virginia General Assembly's creation of the Southwest Virginia Economic Development Commission, which was established to improve social and economic development in the region. Much of the commission's focus has been on improving the transportation infrastructure (e.g., Virginia State Highway 58), providing assistance and incentives to business, marketing southwest Virginia, and developing natural resources. Industrial park expansions, landfills, prisons, and airports, as well as construction of a new major highway transecting the area, are proposed in the watershed.

The karst areas may be impacted if plans for a new airport and prison are implemented. It appears that the prison will be constructed in an area and manner that will minimize impacts to the karst system. FWS, the Virginia Division of Natural Heritage, and the Virginia Cave Board will be working closely with a number of partners to mitigate negative impacts from the proposed airport.

Nonpoint-source pollution from urbanization is probably a very important factor in riparian zone modification. It may contribute to elevated temperature, embeddedness, scouring, depositions, and other instream habitat effects. The effects of urbanization were quantitatively analyzed in this risk assessment.

3.2.4 Agriculture—Livestock and Pastureland

Stressors: Toxic chemicals, pathogens/nutrient enrichment, sedimentation, riparian zone modification, habitat destruction

The Bi-State Task Force report to the governors of Tennessee and Virginia identified nonpoint-source pollution as the single most important source of water pollution in the Clinch and Powell watershed. Much of this pollution can be attributed to the poor agricultural practices used throughout the watershed, including overgrazed pastures on steep slopes, animal waste from feed lots, and livestock access to streams and riparian corridors.

Approximately 175,000 acres of pastureland with greater-than-acceptable soil loss tolerances (based on Soil Conservation Service tolerance criteria) occur in the Clinch and Powell watershed. Almost 75,000 head of livestock, mostly cattle, graze in the watershed. The majority of these livestock depend on the river or other perennial streams for water, creating a situation in which degradation of riparian corridors, along with increased nutrient, bacterial, and viral input to the waterways, is common.

Runoff from steep grazing lands may be significant, but the actual extent of sedimentation caused by this type of runoff is unknown. Likewise, the total amount of erosion of streambanks and organic waste pollution resulting from cattle access to streams has not been measured. However, it continues to be significant at a number of sites within the Clinch and Powell watershed. The extent of instream habitat destruction caused by cattle trampling the streambeds has not been determined. Effects of pastureland were quantitatively analyzed in this risk assessment.

3.2.5. Agriculture—Row Crop

Stressors: Toxic chemicals, sedimentation

About one-third of the land in the Clinch and Powell watershed is devoted to agriculture. The extent of pesticide runoff in the watershed is unknown, but tobacco plot-associated toxic chemicals are known to affect karst habitats.

Sedimentation caused by runoff from agricultural lands is expected to have a large impact, especially on benthic organisms in the area. In addition, sedimentation changes cave stream substrates and affects invertebrate habitats. Much of the land that is flat enough to be

used for row crops is contained within floodplain areas. Many natural resource agencies and groups are working to encourage better farming practices that include greenways to reduce runoff, but much of the floodplain is still being cultivated without greenways. Effects of row crop agriculture were quantitatively analyzed in this risk assessment.

3.2.6. Point-Source Discharge—Industrial

Stressor: Toxic chemicals

There are currently 34 industrial point-source discharges within the Clinch and Powell watershed, exclusive of coal-related discharges. The majority of these discharges are associated with small businesses and are classified as minor. Only two major industrial discharges are present in the watershed, one at Foote Mineral on Stock Creek, Scott County, VA, and the other at the Appalachian Power Company's Clinch River Plant (APCO), a coal-fired power plant located at Clinch river mile 267.5. The cumulative impact of these point sources is poorly understood.

Approximately 960 tons of fly ash is produced daily at the APCO coal-burning power plant as a result of the high ash content of the coal used at the plant. Water withdrawn from the Clinch River is used to transport the ash in a slurry to large lagoons, where the ash settles. The APCO plant discharges various contaminants to the rivers, including copper (Cu), which is especially toxic to molluscs. Cooling tower blowdown effluent averaged 857 μg Cu/L (3–7 μg Cu/L, ambient) in 1977–87. Condenser pipe replacement in 1987 reduced Cu discharge to 100–150 μg Cu/L. The plant, under order from the Virginia Department of Environmental Quality (VDEQ), has been retrofitted to further reduce copper concentrations in the effluent. The new copper standards for the plant are 12 μg Cu/L. After the initial reduction to 150 μg Cu/L, snail recovery was seen 2 years later at a research station 0.9 km below the discharge (Reed, 1993), but molluscan recovery has been much slower (Stansbery et al., 1986). The ability of the new standard to protect the aquatic ecosystems has not yet been validated. It is not known whether unregulated toxic chemicals may be discharged at this and other industrial facilities in the watershed.

The Cypress Foote Mineral plant discharged various contaminants to Stock Creek above Speer's Ferry on the Clinch River. Tests in Stock Creek by VDEQ verified impairment to the creek from the plant and that the discharge may have been contributing to mussel declines at Speer's Ferry. Cyprus Foote Mineral has closed its plant on Stock Creek. However, underground seeps from mine tailings are still contributing high concentrations of lithium and aluminum.

Furthermore, industrial facilities such as the APCO plant have been responsible for catastrophic toxic spills that resulted in severe ecological effects (see Catastrophic Spills, below). Point-source locations were identified and considered qualitatively in this risk assessment. Where point sources were associated with urban or barren land cover (a common phenomenon), urban effects captured at least some of the effects of point sources.

3.2.7. Point-Source Discharge—Municipal Sewage

Stressors: Toxic chemicals, pathogens/nutrient enrichment

Currently, 119 municipal discharges are within the watershed. These include discharges from all the major municipalities in the Clinch Valley as well as from treatment plants at active mining sites. Most municipalities are now in the process of upgrading to secondary treatment standards. Final upgrades have been completed at Richlands and Tazewell on the Clinch River, Pennington Gap on the Powell River, and Coeburn/Norton/Wise on the Guest River. Some extremely rural areas, such as St. Charles on the Powell River and Dante/Hamlin/Castlewood on the Clinch River, continue to discharge raw sewage to the rivers. Pathogens originating from poorly treated municipal wastewater or from failed septic systems and leaking sewers have been shown to cause deleterious effects on fertilized ova (eggs) in the marsupia of female mussels, thus affecting reproduction (Fuller, 1974). However, the extent of this stressor in the watershed is unknown and thought to be insignificant.

VDEQ has banned the use of halogen compounds such as chlorine for disinfection at municipal treatment plants because of the toxicity of halogens to aquatic life. A few treatment plants have not yet been retrofitted with alternative disinfection systems and continue to use chlorine as a disinfectant, with dechlorination mechanisms in place to treat the effluent. A failure of the dechlorination apparatus and subsequent discharge of chlorine from these plants poses an enormous threat to the river in areas such as Cleveland, VA, where an exemplary mussel community lies immediately downstream of the plant outfall. Cleveland has now been upgraded to ultraviolet disinfection. Municipal point sources are associated with urban areas by definition and were therefore implicitly analyzed along with urban effects in this risk assessment.

3.2.8. Silviculture

Stressors: Sedimentation, degradation of riparian areas

This region was cleared extensively upon European settlement and during later migration of people into the area in the late 18th and early 19th centuries. Clearing was driven by the need for agricultural production—both commodity grain and tobacco crops—and for grazing land for cattle and sheep. Logging, with large crews and supported by railroad and tramway systems,

was conducted at the turn of the 20th century to sustain American industrial growth and to salvage lumber from the American chestnut when this tree was devastated by the chestnut blight. The wood- and coal-burning engines used by the logging crews sparked devastating forest fires, which were not adequately controlled for decades—until the 1930s. As logging declined and fires were aggressively suppressed by State and Federal agencies, forests began to regenerate. This recovery was supplemented with the creation of the Jefferson National Forest, whereby large Federal land holdings within several ranger districts were, and remain, dedicated to forest management and protection of forest resources. Today, the region experiences the greatest forest fire risk and activity in the State, but annual losses average less than 5,000 acres per year in the watershed owing to enforcement of the Virginia 4 PM Burning Law, other associated fire laws, and one of the most successful forest fire prevention efforts in the nation.

Harvesting of forest resources continued throughout the 20th century, supplementing the mining industry with timber supports, providing quality hardwoods to furniture and dimension lumber production, and pulp for fine papers to a Kingsport, TN, paper mill. During the 1980s and 1990s, new industries and the availability of former mining industry workers to harvest lumber prompted a resurgence of forest industry in the region. Establishment of an oriented strand board plant in Dungannon (Scott County), VA, has particularly influenced great increases in annual acres logged in several counties of the Clinch and Powell watershed. New forest industries in West Virginia, Kentucky, Tennessee, and Virginia will continue the trend of increased logging of the region's forest resources.

In a 1995 analysis of potential nonpoint-source pollution in Virginia's 493 hydrologic units, performed in concert with soil and water conservation districts and the U.S. Department of Forestry, 20% of the units were assessed as having a "high" potential for nonpoint-source pollution from forest harvesting. Rankings were determined in part by topography and current logging activity. Of the 24 hydrologic units located within the Clinch and Powell watershed, 15 were ranked "high" for nonpoint-source pollution potential from logging, eight ranked "medium," and one ranked "low." Nonpoint-source pollution potential from logging includes erosion and sedimentation, with lesser impacts from petroleum contamination from log decks and areas where equipment is concentrated. When riparian areas are logged, removal of shade also impacts water quality and aquatic resources through increased water temperatures and declines in dissolved oxygen. Removal of sources of detritus and woody debris can also negatively impact aquatic habitats. State BMPs specifications require retention of at least 50% of the basal area in designated streamside management areas. These areas consist of a minimum of 50 feet on either side of the stream, with increasing widths based on topography. A two-zoned riparian buffer is currently under review, in which areas immediately adjacent to the stream would be more rigorously protected.

The U.S. Department of Forestry enforces the Silvicultural Water Quality Law, which can penalize loggers, landowners, and forest producers if potential and actual nonpoint-source pollution from silvicultural operations is not addressed via a system of informal recommendations, reviews, stop-work orders, and hearings. Since passage of the law in 1992, enforcement in southwest Virginia has been aggressive and is supplemented with continual formal and one-on-one educational efforts targeted at loggers, landowners, and forest products producers. As silviculture is still a very minor activity in the watershed, we did not analyze this stressor in this risk assessment.

3.2.9. Hydrologic Changes

Stressor: Habitat destruction

No significant manmade hydrologic changes are known to have occurred or are planned upstream of Norris Dam in Tennessee. Before completion of Norris Dam in 1936, the Powell and Clinch rivers were historically free-flowing, merging with the Tennessee River and then with the Ohio River, and eventually reaching the Mississippi River. Now these rivers flow into the Norris Lake impoundment at Norris, TN, and are thus isolated from the rest of the drainage system. The impact of this type of isolation on the rivers is unknown. Although it is reasonable to assume that some loss of species exchange has occurred because of Norris Lake, this is believed to be a relatively minor source of stress. Therefore, this stressor was not evaluated further in this risk assessment.

3.2.10. Introductions and Migrations of Nonnative Species

Stressors: Competition, infection

The Clinch and Powell watershed, like most natural systems, has been invaded by non-native species. The first aquatic nonnative mollusc species recorded was the Asiatic clam (*Corbicula fluminea*), first seen in the watershed in the 1970s. A second invader, the zebra mussel (*Dreissena polymorpha*), is presently found throughout the mainstem Tennessee River and the lower half mile of the French Broad River. Little is currently known about the impacts of, or controls for, these organisms. Researchers at Virginia Tech are currently studying impacts of *Corbicula* on native mussel populations.

The Asiatic clam was first discovered in the Tennessee River in 1959 (Gardner et al., 1976). Since that time, it has become widespread and extremely common throughout the Clinch and Powell watershed. It is the most common mollusc species in the region. Competitive interaction between the Asiatic clam and native mussel fauna is still not clearly understood, and further research is required.

The zebra mussel, native to the Black and Caspian seas, has spread extensively throughout the Great Lakes region since its discovery in Lake Erie in 1988 (O'Neil, 1991). It has caused devastating impacts to industrial and municipal intakes, natural food chains, and commercial and recreational fishing. No effective means have been developed to control this species, and much concern has been raised about its potential negative impacts on native mussel fauna in the Clinch and Powell watershed. FWS has predicted that as many as 10 species of mussels found in the watershed are likely to become extinct with the establishment of the zebra mussel. However, thus far, the zebra mussel has not been able to establish populations in flowing rivers and streams, because the larvae require more lentic conditions.

Little is known about the impacts of introduced fish on native fish.

We did not analyze this stressor in this risk assessment because of a general lack of information. This may be an important stressor to consider in future phases.

3.2.11. Recreation

Stressor: Nonnative species, overexploitation

Aggressive, nonnative species such as the Asiatic clam and zebra mussel may be spread through the watershed via recreational boating, but the likelihood is not known. There is little boating in the watershed because there are few deep-water reaches. Some whitewater canoeing occurs in the watershed, but it is not extensive. The extent of recreational fishing on the native fish populations is unknown but is considered to be very minor, given the low population density. Therefore, this stressor was not considered further in this risk assessment.

3.2.12. Other Biota—Predation

Stressor: Overexploitation

Increasing populations of muskrats and other predators, such as racoons (*Procyon lotor*) and map turtles (*Graptenys*), prey on hundreds—and perhaps thousands—of molluscs in the watershed each year. The relative impact of predation, although not well documented, may be a significant threat to mussels because of the abundance of muskrats in the watershed. In addition, muskrats tend to select smaller species of mussels, which, in many cases, are the most endangered species. Observations by some scientists indicate that muskrat predation appears to be inhibiting the recovery of endangered mussel species in the Clinch and Powell watershed and is likely placing some populations of the endangered pigtoe mussel (*Fusconaia edgariana*) in jeopardy of extirpation. Information about this stressor is severely lacking and, therefore, could not be analyzed in this risk assessment. This may be a stressor to consider in future phases.

3.2.13. Illegal Harvesting

Stressor: Overexploitation

Earlier in the 20th century many mussels were harvested for the button industry. Today that type of harvesting is illegal, but it still occurs to an unknown extent. This is believed to be a very minor source of stress in the watershed, and it is unlikely to become major, given the low population density in the watershed and much larger, legal sources of mussel harvesting in other basins (e.g., Ohio River, Mississippi River). Therefore, this source of stress was not further evaluated in this risk assessment.

3.2.14. Catastrophic Spills

Stressors: Toxic chemicals, sediments

Catastrophic spills of toxic materials into surface waters of the Clinch and Powell watershed have long been recognized as a major water quality stressor. Past spills have been high-magnitude, short-duration, low-frequency events that undoubtedly caused extensive long-term impacts on native species in the basin. Because of the unpredictable and episodic nature of these spills, we were unable to quantitatively incorporate this stressor into the risk analyses. However, we made attempts to evaluate catastrophic spills by analyzing effects of proximity to transportation corridors and mining, both of which have been major sources of spills. We also considered spills qualitatively in risk characterization.

The 1996 Virginia Water Quality Report identified six known fish kills, ranging in size from 11 to 11,355 fish. Four kills were the result of accidental spills of cement during construction activities. Historical examples include an October 1993 42-car coal train derailment, which resulted in 4,200 tons of coal being spilled adjacent to the Clinch River at Dungannon, VA. This spill was not reported to VDEQ for several days, and cleanup of the site required several weeks.

The APCO plant has been responsible for two large episodic events that affected Clinch River ecological communities. In June 1967, 440 acre-feet of caustic ash poured into Dump's Creek and then into the Clinch River. For 4 days, the slug of ash traveled downstream, killing all fish it encountered in the vicinity of Carbo and many more for 66 miles of the Clinch River in Virginia and 24 miles in Tennessee. The alkaline excursion was reported to be responsible for eliminating bottom-dwelling fish-food organisms for approximately 5–6 km and snail and mussel populations for 18 km (Cairns et al., 1971). Approximately 216,600 fish were killed in Virginia and Tennessee by the episode. Snails and mussels were eliminated for almost 12 miles downstream.

Studies conducted by Virginia Tech showed that fish and aquatic insects became reestablished relatively quickly following the spill (Crossman et al., 1973). Insect communities showed downstream recovery (i.e., further downstream stations had higher density and diversity)

in 1969, but molluscan communities had not recovered for at least 30 km below the spill site. Presently, mussels have not yet recolonized the 9- to 10-mile portion of the river below the plant. Differences in invasion and colonization potential between the two groups of organisms underscore the importance of monitoring molluscan populations, because they are slower to recover.

In 1970, before molluscan populations had recovered to prior density, an acid spill occurred at the APCO plant. The area affected was less extensive than that of the fly ash spill: 13.5 miles downstream to St. Paul, VA. Approximately 5,300 fish were killed (Crossman et al., 1973). After the spill, no surviving mayfly or molluscan species were found for 18 km below the spill. Within 6 weeks, diversity of arthropod benthic organisms had recovered, but mussel species had not.

The potential for future spills is not clear. A catastrophic spill can originate at industrial facilities located along the rivers or from accidents along transportation corridors that cross or parallel the waterways and karst systems. Additionally, catastrophic spills can result from illegal dumping into waterways or sinkholes. These spills can pose a potentially enormous threat to the riverine ecosystem, as previously described. It is acknowledged that many spills of toxic chemicals may have occurred in the past and been unreported.

Unfortunately, little information has been accumulated on storage and transportation of toxic materials in the basin, and the full potential for impacts to the fauna is unknown. Consequently, the development of contingency plans will be limited until additional information on toxic material transport and storage can be obtained.

3.3. SIMPLIFIED CONCEPTUAL RISK MODEL

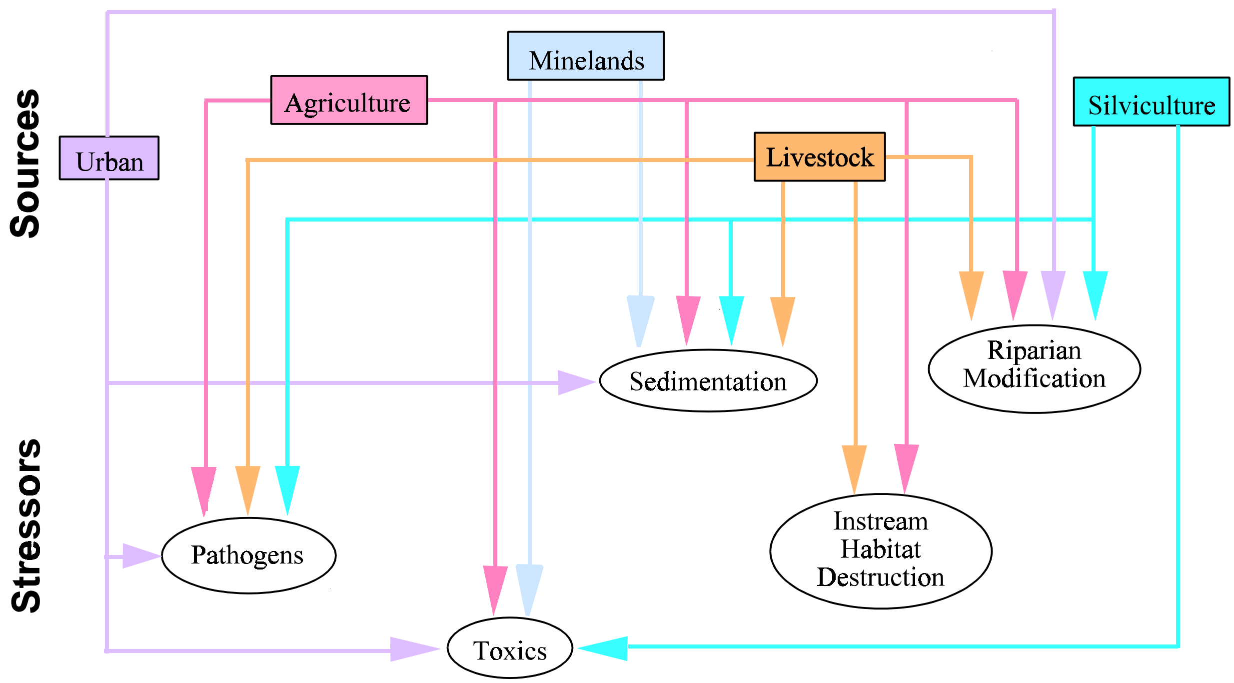
The conceptual model describes pathways between human activities (sources of stress), stressors (which may be physical, chemical, or biological in nature), and assessment endpoints (U.S. EPA, 1998). The model yields predictions or risk hypotheses of how human activities affect the valued ecological resources (assessment endpoints) and is based on ecological experience and best professional judgment. The two conceptual models, one for each assessment endpoint analyzed in this risk assessment, were developed by the workgroup as part of problem formulation and are described below and in Figures 3-3 and 3-4. The models shown do not portray all possible sources and stressors and the pathways by which they might impact ecological resources within the watershed. Developing and presenting such a comprehensive model was found to be neither helpful nor resource-efficient in the context of this risk assessment. Only those pathways or relationships that were considered most ecologically important in this region by resource managers are depicted in Figures 3-3 and 3-4. These more simplified conceptual models were very useful for identifying potential data sources (and data

gaps) and tracking progress in specific risk analyses. Stressors and their sources were grouped into three major categories: degraded water quality, physical habitat alteration, and biotic interactions (Table 3-1). The following sections briefly discuss known or assumed effects of these different types of stressors on assessment endpoints in this risk assessment.

3.3.1. Degraded Water Quality

Host fish survival and reproduction are affected by several of the same sources and stressors as mussels (see Figure 3-3), thus accentuating these stressor effects on mussels. Toxic chemicals such as heavy metals or chlorine, which are discharged by some municipal and industrial wastewater dischargers in the watershed, and pesticides originating from agricultural activities in the watershed are known to have severe effects on mussel survival and recruitment (Havlik and Marking, 1987; Sheehan et al., 1989; Goudreau et al., 1993; Reed, 1993). Mine water discharges may contain other pollutants, such as hydraulic oils, foaming agents, surfactant materials, and greases that can be extremely toxic to filter-feeders such as mussels (BMI, 1990; Lingenfelser, 2000). These pollutants may enter the stream via surface water discharges or underground springs and caves that surface somewhere else in the watershed. Thus, mussels and fish may be affected by subsurface as well as surface water quality. Urban stormwater runoff and untreated or failing septic system waste also may release pollutants, compounding toxic stress to mussels and fish. The fact that mussels are sedentary and benthic makes them even more of a target for water pollutant exposure (Sheehan et al., 1989). The siphoning mode of feeding used by mussels also makes them susceptible to bioaccumulative effects of organic pollutants (Tessier et al., 1984; Kauss and Hamdy, 1991; Livingstone and Pipe, 1992).

Figure 3-3. Conceptual risk model for mussels.



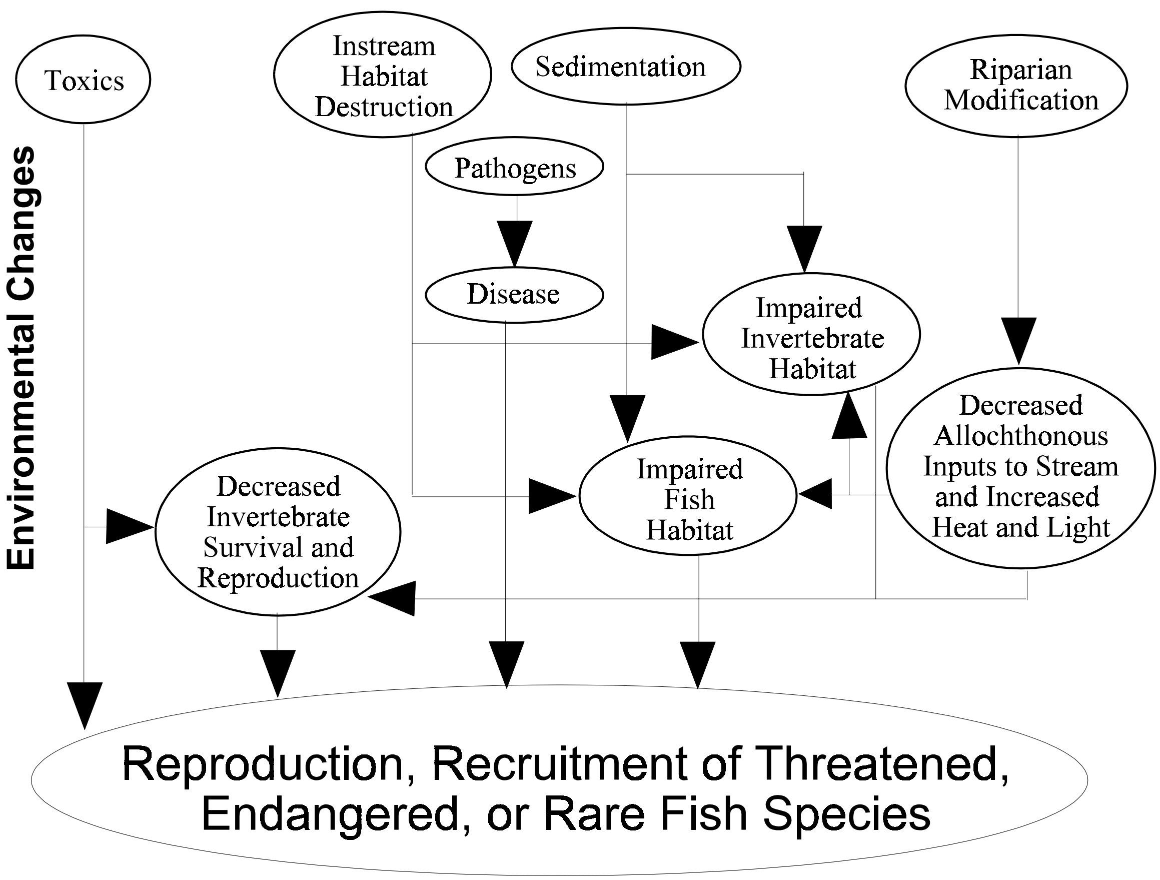


Figure 3-4. Conceptual risk model for fish.

3.3.2. Physical Stream Habitat Alteration

A variety of activities in the watershed could result in deterioration of instream benthic habitats. Any activity resulting in increased sediment deposition and substrate embeddedness instream reduces the amount of available benthic habitat necessary for successful mussel larval settlement, growth, and survival (Bates and Dennis, 1978; Way et al., 1990) and fish spawning and feeding habitat (Freeman, 1987). This in turn directly affects recruitment of mussel and fish populations. Thus, soil runoff resulting from poor agricultural practices, livestock trampling instream, elimination of the riparian corridor, and urbanization could directly influence the amount of available habitats for mussels and fish.

Significant alterations in stream flow or channel modification are other stressors that could directly affect mussel and fish habitat availability and, therefore, mussel and fish diversity and abundance. In addition to large-scale human activities that can severely modify aquatic life habitat, such as dams and dredging, more subtle forms of habitat alteration may be important in the Clinch and Powell watershed, such as (1) riparian corridor and stream bank destabilization due to transportation corridors and poor agricultural and silviculture practices, (2) high-flow regions and benthic scour due to large industrial and municipal wastewater discharges and urban stormwater runoff, and (3) bank failure, channel widening, and subsequent channel depth and current velocity reduction due to livestock watering instream. Turbidity resulting from sedimentation or livestock wading and watering instream further affects mussel and fish survival and recruitment by interfering with, or reducing the effectiveness of, normal feeding and larval survival (Stansbery and Stein, 1976; Dennis, 1981, 1985).

Another major stressor in the form of physical habitat alteration is the loss of the riparian corridor. The workgroup recognized that the magnitude of various stream effects (e.g., water quality or physical habitat alterations) on stream biota is likely to be a function of the riparian corridor integrity present. Therefore, loss of riparian corridor warrants separate and more extensive discussion.

3.3.3. Loss of Riparian Corridor

The composition and connectivity of riparian vegetation are potentially affected by several different human activities in the watershed, including livestock grazing, agricultural row crop, forestry, mining, silviculture practices, urban development, wastewater discharges, transportation corridors, hydrologic modification, and perhaps acid rain deposition (Minshall, 1993; Richards and Host, 1994). Most of these activities result in thinning or removal of the natural riparian vegetation (particularly the canopy). This alteration could have several effects, including loss of soil and nutrients, soil instability, and bank erosion or failure (Cooper et al.,

1987), thereby altering the species composition and connectivity of the riparian corridor as a whole. Each of these stressors directly affects the abundance and composition of plant species capable of inhabiting the riparian zone and, ultimately, channel stability. Loss of soil and bank failure affect sedimentation instream, which is a major potential stressor for the assessment endpoints examined in this risk assessment.

Lack of an intact and connected riparian corridor is expected to reduce mussel and fish populations through other habitat-related stressors. Riparian canopy removal or thinning increases light and heat penetration to the stream bank, resulting in higher temperatures and lower dissolved oxygen saturation instream, both of which could be deleterious to mussel and many fish species. Riparian vegetation also captures sediment from overland flow, retards floodwater, and captures fertilizer and other chemicals in runoff from agriculture fields (Cooper et al., 1987; Osborne and Kovacic, 1993; Richards and Host, 1994). Also, removal of the natural riparian corridor (either from agricultural, urban, transportation, or forestry activities) reduces or eliminates the important exchange of nutrients and allochthonous energy between the stream and its floodplain, resulting perhaps in reduced food availability for mussels, other invertebrates, and fish (Gregory et al., 1991).

Removal of riparian forests can greatly diminish sediment and nutrient trapping capabilities of areas immediately adjacent to streams. Certain aquatic invertebrate species have declined in other systems because there are fewer riparian refugia during floods and other periods of environmental stress and because of a reduction in detritus and woody debris that historically served as the major energy source (Minshall, 1993). Thus, riparian corridor alteration is treated as a stressor in this risk assessment in the sense that stream habitat quality and perhaps water quality could be degraded if riparian corridor connectivity is impaired by human activities.

3.4. ANALYSIS PLAN

The goal of risk analysis is to draw meaningful and statistically supported relationships based on available data. Initially, the relationships presented in the conceptual models (section 3.3) formed the basis of hypotheses that we planned to analyze using data collected in the watershed over many years. This risk assessment initially assumed that certain stressor and effects data would be available from different subwatersheds. As the workgroup began to explore different hypotheses in this risk assessment, it modified the analytical approach in accordance with the data actually available and the results of initial analyses. For example, water chemistry data were lacking in this assessment, which limited our ability to analyze relationships between water quality stressors and either land uses or biota. It was ultimately decided to evaluate the following risk hypotheses:

Physical Habitat Alteration Hypotheses

- Greater riparian connectivity or forested riparian vegetation is associated with greater diversity and abundance of mussels, other macroinvertebrates, and native fish.

- Watershed areas dominated by agricultural, urban, or mining land uses are associated with poorer habitat quality and biological diversity than are forested or naturally vegetated areas.

Water Quality Hypothesis

- Proximity to nonpoint-source runoff from agricultural activities and urban areas and point-source discharges (including coal mining discharges) result in detrimental structural changes to native mussel and fish populations.

3.5. ANALYTICAL APPROACH

To test these hypotheses, the general analysis scheme entailed identifying potential patterns or relationships between different land uses and stressor measures. Relationships between land-use activities and measures of effect representing the assessment endpoints were also examined. Interpretation of the results from these two sets of analyses allows inferences to be made about the relationships between specific stressors, or combinations of stressors, and assessment endpoints.

The objectives of the analysis phase are to gain a better understanding of (1) the extent to which ecological resources are exposed to stressors resulting from human activities and (2) the likely effects that may occur. These exposure and effects characterizations are then integrated into an overall estimate of risk in the next phase, risk characterization.

The analysis of risk in this assessment relies on current and past land-use practices and measurements taken at specific sites in strategic subwatersheds in the Clinch and Powell watershed. Past stressors and effects were then used to evaluate future risks of similar sources and stressors in other parts of the watershed. Watershed risk assessments are complex because of the co-occurrence of stressors and multiple pathways by which stressors impact assessment endpoints. The multiple sources, stressors, and pathways and their co-occurrence at such a large spatial scale lead to greater uncertainty and reduced associations among specific sources, stressors, and ecological responses. The complexity of the assessment is further compounded by the lack of an optimal suite of data.

Furthermore, there may be problems combining data from many sources, especially if collected for another purpose (ITFM, 1995; MDCB, 1999). The technology is not yet available to develop associations at such large spatial scales between multiple sources (e.g., agriculture, forestry, and urbanization), stressors (e.g., impaired water quality, sediments, and toxic substances), and observable biological effects; the data requirements are enormous as well. Consequently, for watershed ecological risk assessments, exposure estimates may need to be aggregated at the landscape level, whereby exposure is inferred from source or land-use information. Alternatively, analyses may need to be limited to the most disruptive stressor. This risk assessment was initiated in part to develop an approach for addressing the complexity of this task, because no precedence exists on how to perform such assessments.

Given these limitations, we first examined relationships between land use and biological data, as both these types of information were relatively reliable and available. Exposure information, (i.e., inferred condition of habitat impacted by various physical and chemical stressors) was based on knowledge from a few sites in the watershed. On the basis of available data in the watershed and information from the literature, we evaluated the effect of nearby land uses on the condition of physical habitat and biota as a surrogate for quantitative profiles of exposure and effects. Thus, the exposure-effects distinction, typically the norm in chemical risk assessments, was not as useful in this assessment because of a lack of adequate stressor data.

The land cover Dataset used in this risk assessment was derived from classified Landsat Thematic Mapper imagery (Hermann, 1996). This Dataset was created as part of the Southern Appalachian Assessment, and imagery was classified into 17 discrete categories using the following scheme:

1. Northern Hardwood Forests
2. Mixed Mesophytic Hardwood Forests
3. Oak Forests
4. Bottomland Hardwood Forests
5. White Pine/Hemlock Forests
6. Montane Spruce–Fir Forests
7. Southern Yellow Pine Forests
8. White Pine/Hemlock/Hardwood Forests
9. Mixed Pine/Hardwood Forests
10. Herbaceous
11. Barren
12. Agriculture–Pasture
13. Agriculture–Copeland

14. Wetlands
15. Developed
16. Water
17. Indeterminate–Clouds, Shadows

For risk analyses, the different forest categories were aggregated into one forest category. All terrain data (e.g., elevation and slope) were derived from a mosaic of 30-meter resolution USGS digital elevation models (USGS, 2001). The stream network data used was EPA's River Reach (RF3) data (U.S. EPA, 2000). Point locations of mines and processing plants came from the U.S. Bureau of Mines MAS Dataset (Causey and Douglas, 1998).

Table 3-2 summarizes available biological and habitat measures of effect used in this risk assessment. The vast majority of these data were collected by TVA staff as part of regular monitoring programs, including the CPRATS (N = 155 sites) and the CMCP (N = 60 sites). The CPRATS sampling program is a fixed-station design that relies on targeted samples stratified by stream size and general area of the watershed. The CMCP program is a targeted sampling design that relies on historic mussel information as well as habitat quality, as judged by field malacologists.

Each of the hypotheses stated in this section required a series of measurement endpoints or metrics for analysis incorporated into a GIS. Critical data on environmental stressors, sources of stressors, and biological resources were mapped to help identify co-occurrences. Source and stressor data layers included land cover, stream drainages (USGS Stream Reach File 3), road density, locations of point-source dischargers and mines, and stream habitat quality indices. Biological data relied on in this risk assessment included the fish IBI, native mussel species richness and abundance, and the macroinvertebrate EPT family index. Most biological and habitat data analyzed in this study were collected between 1980 and 1996, although some historic data (pre-1920) were also used. Some data were also obtained from FWS sampling records for threatened and endangered species and from papers published by other researchers.

Water quality data (e.g., concentrations of toxic chemicals in effluents and streamwater) were fairly limited in both spatial distribution and extent of information. Few priority pollutants have been routinely monitored in this watershed. Therefore, we relied on available conventional pollutant data derived from EPA's Storage and Retrieval System (STORET), as analyzed by Zipper et al. (1991); discharge reports for the major industrial and municipal wastewater facilities; and available information contained in EPA's BASINS (Version 2.0) database, which includes some relevant data in EPA's permit compliance system.

Table 3-2. Available data used in risk analysis

Riparian corridor and instream habitat quality	Invertebrates	Fish
Substrate embeddedness Epifaunal substrate Riparian forest cover Channel width Channel depth Floodplain width Vegetative cover Bank integrity Sediment erosion rate Instream cover Substrate particle size Presence/absence of wood vegetation Habitat quality index	Mussel species presence/ absence Mussel size classes[a] Mussel species abundance[a] Benthic macroinvertebrate EPT Native mussel species richness Threatened and endangered mussel species richness Pleurocerid snail species abundance[a] Sediment toxicity to mussels (various species and life stages) and other invertebrate fauna[a] Aquatic acute and chronic toxicity to mussels (various species and life stages) and other invertebrate fauna[a]	Threatened and endangered fish presence/absence Fish IBI scores Habitat suitability indices

[a]Data were available for a limited number of sites within the watershed.

The IBI is an index of fish community integrity that is composed of 12 different metrics (in the version used) or components of the fish community, ranging from individual-level characteristics (e.g., incidence of tumors or lesions) to community-level characteristics (e.g., percentage of sunfish or darter species or specific feeding guilds) (Karr and Chu, 1999). The IBI is scored so that one can discriminate among sites that have poor, fair, good, or excellent fish community integrity. The scores are derived by comparison with selected reference sites within the watershed being characterized (Barbour et al., 1999; Karr and Chu, 1999). Reference sites represent the best conditions existing at the time and serve as a control or baseline. The individual scores are added and the composite scores are then grouped into integrity categories (poor, fair, good, or excellent), based on comparison with the reference site. Sites with IBI scores corresponding to poor or fair condition indicate an impaired or stressed fish community; IBI scores corresponding to good or excellent condition indicate an unimpaired fish community. Although it would have been useful to examine the 12 separate IBI measures (Norton et al., 2000) in addition to the composite scores, at the time of this study only the composite IBI scores were accessible for analyses. Use of only the composite IBI scores potentially introduced uncertainties in our risk analyses, as discussed later in this report.

The EPT is an index representing the number of macroinvertebrate taxa belonging to the taxonomic orders *Ephemeroptera* (mayfly), *Plecoptera* (stonefly), or *Trichopera* (caddisfly).

These three orders of macroinvertebrates are generally thought to be relatively more pollution-intolerant than other taxonomic orders (Lenat, 1984).

The fish IBI and macroinvertebrate EPT measures are believed to have a fairly high level of confidence because, by and large, most streams in this watershed are wadeable during normal flow conditions, affording relatively high sampling efficiency. Furthermore, both types of fauna (as well as mussels) have been sampled for more than 30 years by TVA biologists, using standard protocols. Macroinvertebrate samples were collected from riffle areas only. Fish were sampled via electroshocking, sometimes from a boat in occasional pool areas.

3.6. PILOT TESTING

We performed preliminary analyses on the effects of different sources and stressors on endemic mussel and fish species in four subwatersheds: Copper Creek, the Guest River, the upper Clinch River, and the upper Powell River. Copper Creek drains into the upper Clinch River and the Guest River drains into the upper Powell River. The four subwatersheds comprise a range of different land uses, particularly differences in coal mining activity, pasture and crop area, and urban area (Table 3-3). Rigorous analyses were not feasible because of the lack of biological data in some subwatersheds. Comparisons of available biological data among subwatersheds and between the Clinch and Powell rivers and analyses of data for the watershed as a whole were used to infer relationships and augment the existing body of knowledge relating sources and/or stressors and native species distribution. The lower segments of the Clinch and Powell rivers were addressed by references to the literature but were not analyzed extensively because of resource limitations.

Before conducting risk analyses for the watershed as a whole, we first pilot-tested our analytical approach in one subwatershed to address two analysis objectives central to this assessment: (1) to identify the appropriate spatial scale to test relationships between land-use activities or stressors and measures of effect and (2) to identify whether the benthic macroinvertebrate measure (i.e., the EPT index) or the fish IBI is a reliable surrogate measure of effect for predicting the status of native mussel assemblages. Achieving the latter objective was especially desirable, because it was known at the outset of this study that available native mussel data were more limited than either EPT or IBI values. Copper Creek was chosen for this pilot analysis because it was the most data-rich subwatershed and because it was a relatively simpler case in that agricultural uses were the major source of anthropogenic activity (Table 3-3).

Table 3-3. Comparison of land cover for four subwatersheds examined in the Clinch and Powell watershed risk assessment

Land Cover	Upper Clinch River	Upper Powell River	Guest River	Copper Creek
Forest (%)	53.7	89.6	84.1	57.7
Copeland (%)	0.6	3.1	<0.1	1.3
Pasture (%)	44.5	2.4	10.4	40.9
Urban (%)	1.1	4.2	2.6	<0.1
Number of mines[a]	8	21	26	0

[a]Mines and coal preparation plants.

To address the first objective, ArcView (v. 3.0, ESRI, Redlands, CA) was used to examine several different stream riparian widths (50, 100, 200, and 400 m) and several different distances upstream of each sampling point (100, 200, 500, 1000, and 2000 m). This analysis was designed to evaluate the optimal spatial scale to determine the relative influence of riparian corridor or valley agricultural activities on resulting biological integrity or habitat quality at a site. Percent agricultural land cover (pasture and Copeland) was then calculated for each different combination of riparian width and distance upstream for each of nine fish and macroinvertebrate sites sampled by TVA between 1995 and 1996 in its CPRATS program. The relationship between percent agricultural land cover and the fish IBI, the macroinvertebrate EPT, and habitat quality indices for the different combinations was determined using Spearman rank correlation (p=0.05). The IBI, the EPT, and the habitat quality index were also analyzed in relation to riparian percent agricultural land cover upstream of each sampling point for the entire subwatershed.

Recognizing there was uncertainty in relying on only Copper Creek data to determine appropriate land-use dimensions for risk analyses, a similar analysis was undertaken to quantify this uncertainty. Mussel data collected in the upper Clinch (Jones et al., 2000) and a subset of IBI data from the CPRATS Dataset were used in these analyses. Section 3.6 describes the specific methods used for these subsequent analyses.

The fish IBI and the EPT were evaluated as potential surrogate mussel indicators (analysis objective 2 above) by regressing either mussel abundance or species richness at 27 sites in Copper Creek (TVA CMCP 1981 data) (TVA, 1981) with calculated fish IBI measures based on data collected by Masnik (1974) at similar locations. The EPT was further evaluated as a surrogate mussel indicator by qualitatively examining mussel abundance and species richness at nine sites that were adjacent to the nine TVA EPT collection sites in Copper Creek.

Results of pilot analyses were used to define the approach for investigating stressors and sources for each biological sampling location in the Clinch and Powell watershed in subsequent

analyses. A coverage was created using ArcView to display the location of potential human sources of stress (e.g., urban centers) along with TVA's CPRATS sites for the entire watershed. With ArcView Spatial Analysis (ESRI), a 2-km area was created around each sampling site to identify the sources present at that site. Data from these coverages were then aggregated into a single table and imported into Statistica for analysis.

Subwatersheds that exhibited low, moderate, or high levels of certain land uses in comparison to others in the basin (Table 3-3) were also compared to determine probable causes of biological responses.

3.7. STATISTICAL ANALYSES

TVA, under the supervision of the workgroup, developed and entered land-use, habitat, and biological data into a GIS using Arc/INFO (v. 7.04, ESRI) and partitioned in various ways using ACCESS (Microsoft) to obtain databases that were amenable to various statistical analyses (Statsoft, v. 5.0, Tulsa, OK) to support analysis of impacts on assessment endpoints identified in this risk assessment.

Relationships between stressors or sources and biological measures of effect were identified using forward stepwise multiple regression analyses ($p<0.05$ for the overall model). In forward stepwise multiple regression analyses, independent variables are entered one at a time to analyze how much each one adds to the explanation of the dependent variable. Independent variables included percentages of various land uses and number of mines in the riparian corridor as well as in the immediate drainage upstream of biological sampling points. Type I error was controlled by limiting analyses to no more than one factor (independent variable) per 10 sites and by including only those variables that increased the overall R^2 by at least 10%. Variables with p values > 0.05 were considered in multiple regression analyses only if their F value was sufficiently high to be entered into the model and if the resulting R^2 value was at least 10% greater. Multiple regression analyses were supplemented with relevant bivariate plots and univariate statistics (e.g., analysis of variance [ANOVA] t-test; $p<0.05$) to confirm regression results.

In addition, some variables such as the IBI, the EPT, and certain land-use data (e.g., distance from mines or urban areas) were categorized in various ways to determine whether nonlinear or categorical relationships were present between sources and biological measures of effect. For each biological sampling point, the effect of proximity to the nearest urban centers, major roadways, or coal mine activities upstream was calculated and categorized as either <1 km, 1–2 km, or > 2 km, based on the preliminary riparian corridor analyses mentioned above. Biological and habitat data were subjected to one-way ANOVA using these three proximity categories as class variables and significance defined as $p<0.05$. Fish IBI or habitat measures

were also categorized on the basis of TVA's biological criteria ratings for unimpounded streams (Angermeier and Smogor, 1993). In general, we analyzed two categories of fish community integrity or habitat quality—impaired or unimpaired—on the basis of TVA's numerical criteria. These categories constituted class variables in a one-way ANOVA in which the independent variables were percentages of various land uses or number of mines upstream of the biological sampling point.

4. PILOT TEST OF RISK ANALYSIS APPROACH

4.1. BACKGROUND

Before implementing the risk analysis plan in the problem formulation, we pilot tested the proposed analytical approach in a single subwatershed to address several specific questions:

- What is the appropriate spatial scale to test relationships between land-use activities and stressors or measures of effect?

- What is the appropriate spatial context to test relationships between riparian corridor vegetation (integrity) and measures of effect?

- What is the relative effect of upland versus riparian land-use activities on stream habitat indices and measures of effect?

- To what extent can stream habitat indices be related to measures of effect?

- Is either the EPT or the fish IBI a reliable surrogate measure for native mussel assemblage measures?

The Copper Creek subwatershed (Figure 4-1) was chosen to pilot-test our analytical approach for several reasons. First, more data on fish, mussels, macroinvertebrates, and habitat measures exist for this subwatershed than for the others. Second, the Copper Creek watershed has relatively little urban or mining influences, compared with other subwatersheds in the Clinch and Powell basin (Table 3-3), which made it easier to interpret effects of varying spatial scale or upland/riparian comparisons on biota. Third, the Copper Creek subwatershed is fairly small (34,344 ha) and therefore relatively manageable in terms of addressing the above questions. Fourth, only this subwatershed had older (1960s to 1970s) and more recent (1990s) fish and mussel data that could be used to interpret the effect of implementing agricultural BMPs in the 1980s. Finally, there were adequate mussel, IBI, and EPT data to determine the degree to which either IBI or EPT values could be used as surrogate measures of effect for the mussel community.

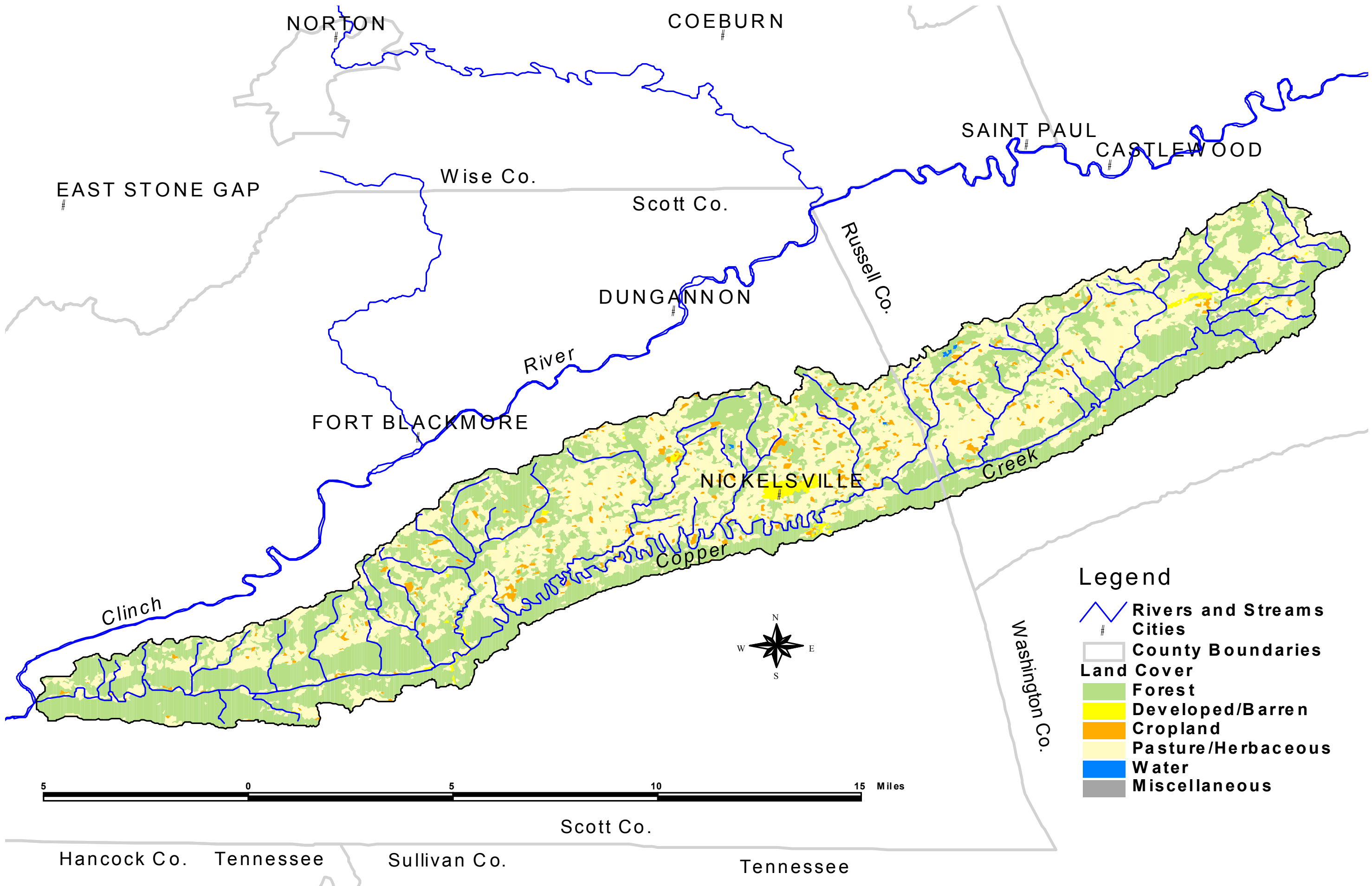


Figure 4-1. The Copper Creek subwatershed and major land uses.

4.2. RESULTS

4.2.1. Riparian Corridor Analyses

Initially, we examined percentages of different land uses upstream of each of the nine CPRATS biological sampling points in Copper Creek. This analysis was used to examine upland land-use effects on stream habitat and biological measures of effect (as defined in Table 3-2). Figure 4-2 summarizes results of this analysis. The fish IBI showed weak negative relationships with percent agricultural (crop and pasture) area upstream, whereas the EPT appeared to be unrelated to upstream agricultural area. TVA's habitat quality index also appeared to be unrelated to percent agricultural land use upstream. Thus, upland agricultural area appeared to have little relationship to measures of effect examined in this analysis.

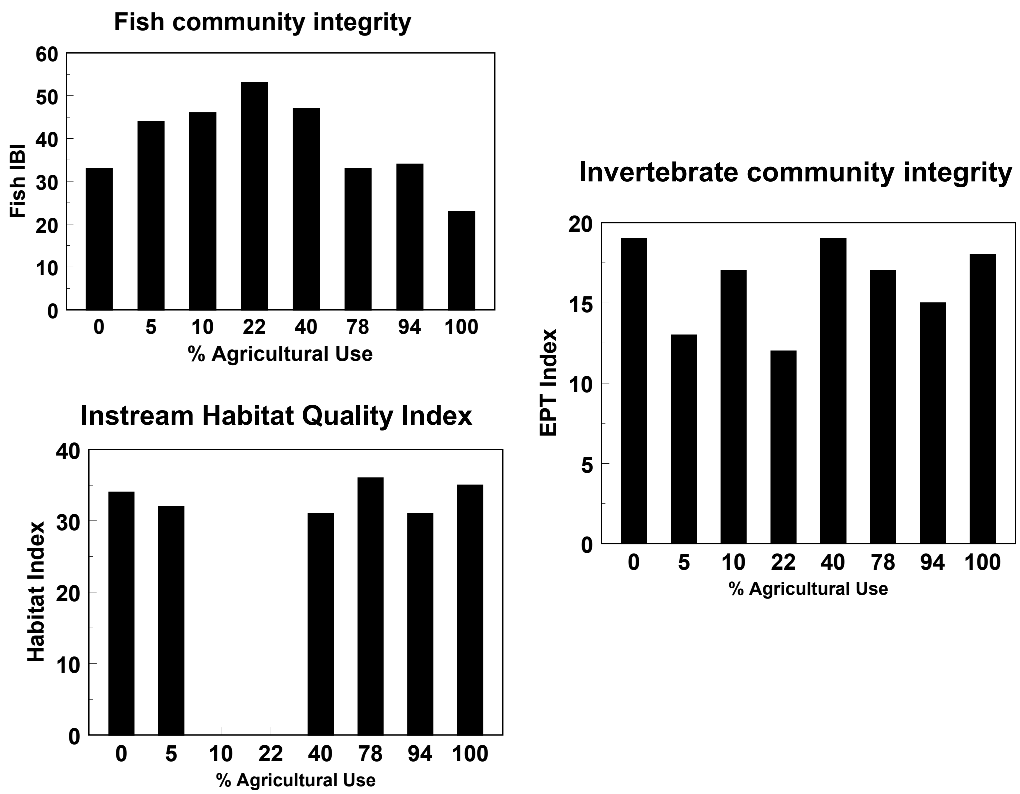


Figure 4-2. Summary of fish IBI, EPT, and habitat quality data for nine TVA sites surveyed in Copper Creek, 1995–96, as a function of total upland agricultural area above the sampling point. Habitat quality data were not collected by TVA at two sites.

We then examined land-use effects at various riparian distances from the stream and different distances upstream. Each biological sampling point was buffered at several different distances surrounding the stream: 50, 100, 200, and 500 m and at 100, 200, 500, and 1000 m upstream. Land-use proportions were then computed for each of these combinations for each

sampling point. We were particularly interested in contrasting the effects of percent forested area or percent pasture area in these different riparian areas and IBI or EPT scores. Relationships were not strong because sample size was limited (N = 9); however, significant correlations between either percent forested area or percent pasture and the IBI appeared to be evident in
intermediate-size riparian areas (200 m buffer width and 500 to 1000 m upstream) (Table 4-1; Figure 4-3). More refined land-use analysis of the CPRATS sites, using a 200 m buffer (100 m on each side of the stream) and a 500 to 1500 m distance upstream, suggested an inverse correlation between percent agricultural land in the riparian corridor and the IBI (Figure 4-4). Analyses based on shorter or longer distances upstream (< 500 m or >1500 m) or narrower or wider riparian widths appeared to result in poorer correlations between land use and the IBI. The EPT was not as closely related to riparian land-use percentages as was the IBI. As many of these sites were between 5 and 15 km away from each other, there was some uncertainty as to whether there may have been cumulative effects of riparian corridor impacts for longer distances upstream of each sampling point. This source of uncertainty was examined further using upper Clinch River mussel data and fish IBI data for the watershed as a whole (see Chapter 5).

Correlation analyses using the nine CPRATS sites (six of which had habitat scores) indicated no significant correlations between either percent upland agricultural land use or riparian agricultural land use and habitat metrics such as embeddedness and instream cover (Pearson correlation analysis, $p>0.05$). However, the sample size was very small.

Results of these preliminary analyses suggest that riparian vegetation characteristics upstream of the sampling point are probably more important than total upland agricultural area in defining native fauna integrity if the riparian area is not too narrowly defined; that is, 100 m on either side of the stream (200-m width altogether) and several hundred meters upstream. This riparian area could constitute a stream-specific optimal riparian management area within which to better prioritize protection efforts (Figure 4-5). The stronger relationships between riparian land use and biota shown with these riparian corridor dimensions suggest that the fauna were responding to influences outside and upstream of the immediate stream reach sampled. Given limited resources, then, optimal benefits to fish, and perhaps invertebrates, would be realized by focusing restoration efforts on the riparian corridor within a 500 to 1500 m distance upstream and 100 m to either side of the stream for the site of interest.

Table 4-1. Summary of Spearman rank correlation coefficients between percent agricultural area and the fish IBI as a function of different combinations of riparian corridor width and distance upstream for Copper Creek[a]

Riparian width (m)	Distance upstream of biological sampling point (m)				
	100	200	500	1000	2000
50	0.09	0.14	0.10	0.06	0.04
100	0.19	0.24	0.26	0.09	–0.11
200	0.14	0.18	0.30*	0.34*	–0.11
400	0.09	0.14	–0.18	–0.20	–0.19

[a] N=9
*$p<0.05$

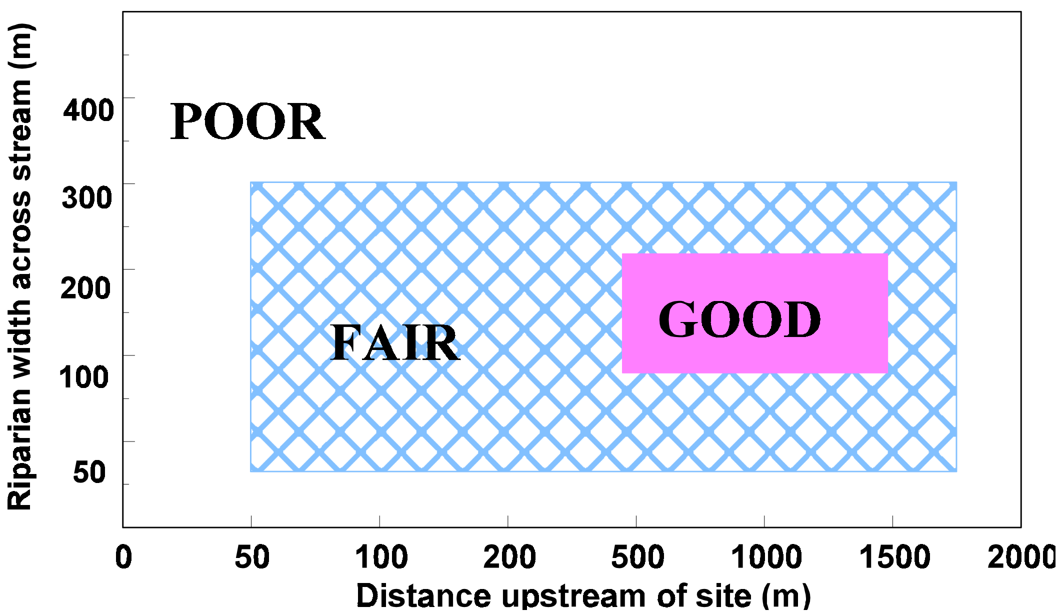


Figure 4-3. Illustration depicting the strength of relationships between land uses and biological measures as a function of riparian buffer width and distance upstream.

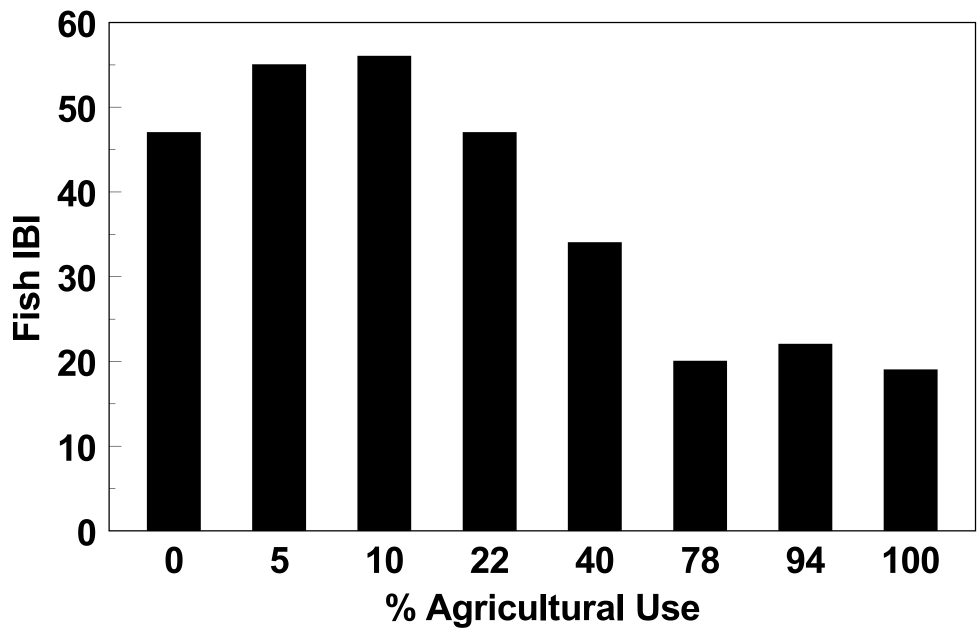


Figure 4-4. Fish community integrity as a function of agricultural land in the riparian corridor (riparian corridor defined as 100 m to either side of the stream and 1,500 m upstream).

Figure 4-5. Some pasture and row crop practices with the floodplain, karst, and lowland areas pose serious risks to aquatic resources.

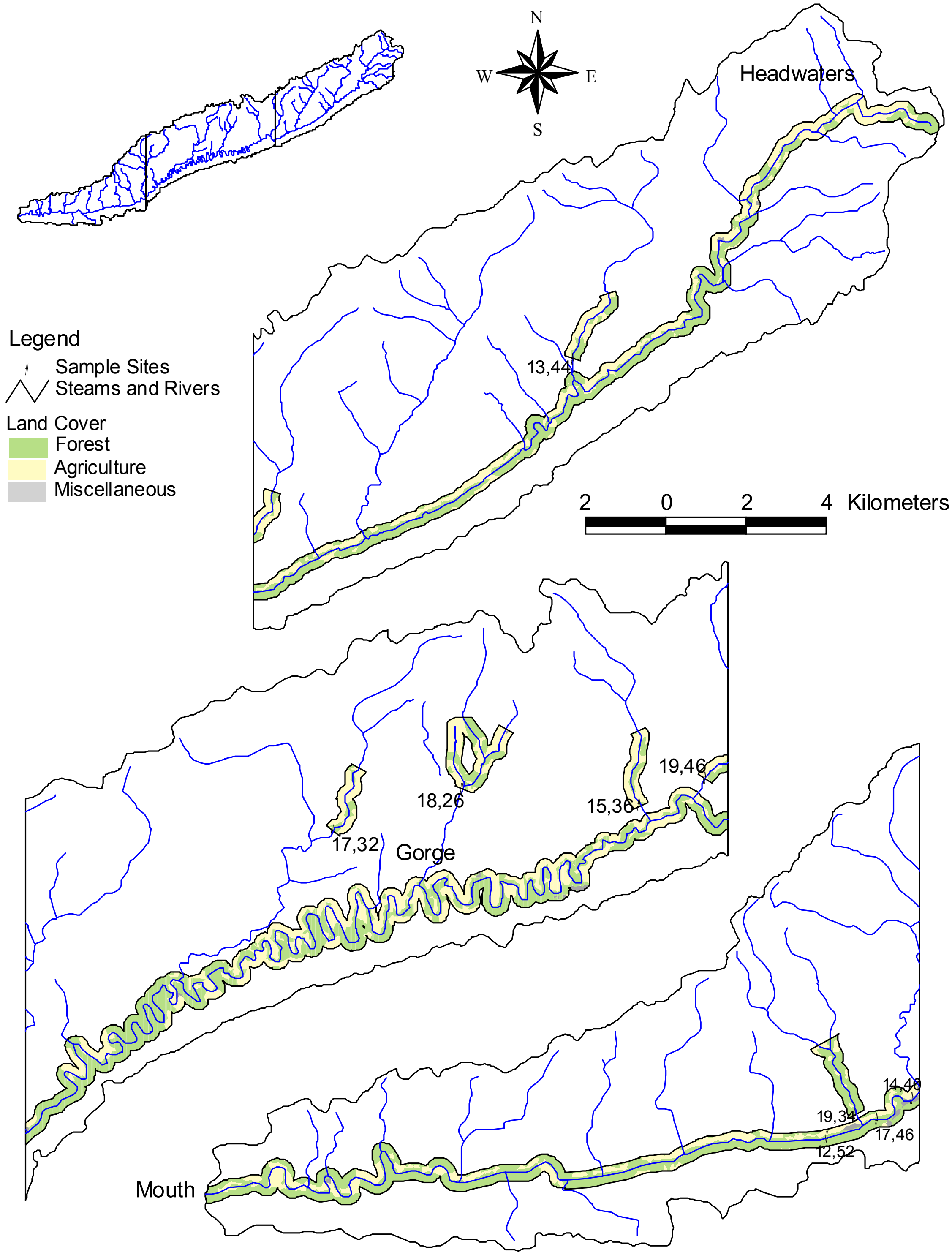


Figure 4-6. Forest and agricultural uses in the riparian corridor (200 m wide) of Copper Creek. Numbers represent macroinvertebrate EPT score and fish IBI values in TVA's CPRATS database, 1995–1996.

We also explored the effect of riparian forest connectivity or continuity on IBI and EPT values in Copper Creek. A 14-acre analysis window was used in which the stream was buffered by 100 m on either side, with each window approximately 1275 m long. Percent forested area was then computed within the 14-acre window. This analysis was repeated for the length of Copper Creek and its major tributaries (Figure 4-6). Comparing IBI and EPT data for the nine CPRATS sites with the forest connectivity information, we estimated that a riparian area consisting of <50% forest or <500 m of continuous forest anywhere within a 1275-m segment had > 75% probability of being associated with impaired fish community integrity. A relationship between the number of threatened and endangered mussel species present at a site and surrounding land uses was also suggested, as shown in Figure 4-7. Riparian areas that were more forested, particularly near the lower part of Copper Creek, exhibited higher numbers of threatened and endangered mussel species than sites with more pastureland cover.

Several lines of evidence suggest that the distribution and abundance of mussels and fish observed in Copper Creek are a function of stressors and not watershed or drainage area. First, historically, many of the headwater areas of Copper Creek and other tributaries to the Clinch and Powell Rivers supported a great diversity and abundance of mussels (Ortmann, 1918) and native fish species (Masnik, 1974). Thus, even small drainage basins in this watershed previously supported diverse and abundant native fauna—far more than what is supported currently. Second, very recent mussel surveys by Ahlstedt (1999) at many of the same sampling stations in Copper Creek showed a decline in mussel species richness and abundance since the last survey, in 1981. Ahlstedt reported increased sedimentation instream at most sites, which is consistent with suggested stressor effects. Finally, even in our current analyses, there are several sites in upstream areas of Copper Creek that have similarly high IBI values as those at sites near the mouth. This observation would be unlikely if watershed area were the driving factor behind the IBI values.

That result suggests that near-field (< 100 m) streamside riparian restoration efforts, for example, might not be an effective means of enhancing fish or mussel diversity. Larger riparian areas (as specified above) would need to be maintained. Furthermore, local instream habitat characteristics may not be related to upland land uses if there is a wide vegetated riparian corridor in those areas. Relationships with family-level EPT were less apparent.

4.2.2. Relationships Among IBI, EPT, and Stream Habitat Measures

On the basis of only the nine EPRATS Copper Creak sites, the IBI was uncorrelated with the EPT $r = 0.50$, p=0.17); however, both endpoints were significantly correlated on the basis of entire CPRATS dataset (N = 95 sites, $r = 0.52$, p<0.01) (Figure 4-8). Neither the IBI nor the EPT was correlated with TVA's overall habitat quality index $r = 0.20$, and 0.22, respectively, p>0.50)

(Figure 4-9); however, both biological indices were correlated with lack of embeddedness r = 0.26, p=0.04 for the IBI and r = 0.29, p=0.02 for the EPT) (Figure 4-9), signifying that substrate quality was a statistically significant feature affecting fish and invertebrate assemblage integrity in the subwatershed as a whole. This analysis also suggests that TVA's multimetric habitat quality index may dampen or mask the effects of specific habitat characteristics on aquatic biological communities. Therefore, for the analysis of the entire watershed, it was useful to also examine relationships between biological measures and individual habitat quality metrics in addition to relationships with the multimetric habitat index as a whole. Analysis of fish IBI measures and embeddedness indicated that fish biological integrity was impaired in 90% of the cases that had high or substantial substrate embeddedness (ANOVA, p=0.03) (Figure 4-9).

In general, embeddedness affects macroinvertebrate and fish assemblages indirectly by reducing the amount of available habitat for shelter, refugia, spawning, egg incubation, etc. As rocks become embedded with fine sediment, the amount of interstitial space available for benthic organisms in the substrate decreases. Thus, significant embeddedness of available substrates will potentially impair the integrity of the biological communities.

In general, the IBI appeared to be a more robust and sensitive measure of effect than the EPT for delineating habitat quality and land-use effects. One possible reason for this result is that the EPT is based on family-level taxonomy. Although the EPT is often sensitive to various habitat and chemical stressors (Lenat, 1984; Diamond et al., 1999), several recent studies have indicated that family-level invertebrate metrics are less sensitive than genus- or species-level metrics to environmental perturbations (Karr and Chu, 1999). Furthermore, the EPT is only one potentially useful metric reflecting macroinvertebrate assemblage integrity (Barbour et al., 1999). Another possibility is that the EPT may recover from spills or other episodic events relatively quickly and, therefore, may not be as sensitive an indicator of past water quality effects as either native mussels or fish.

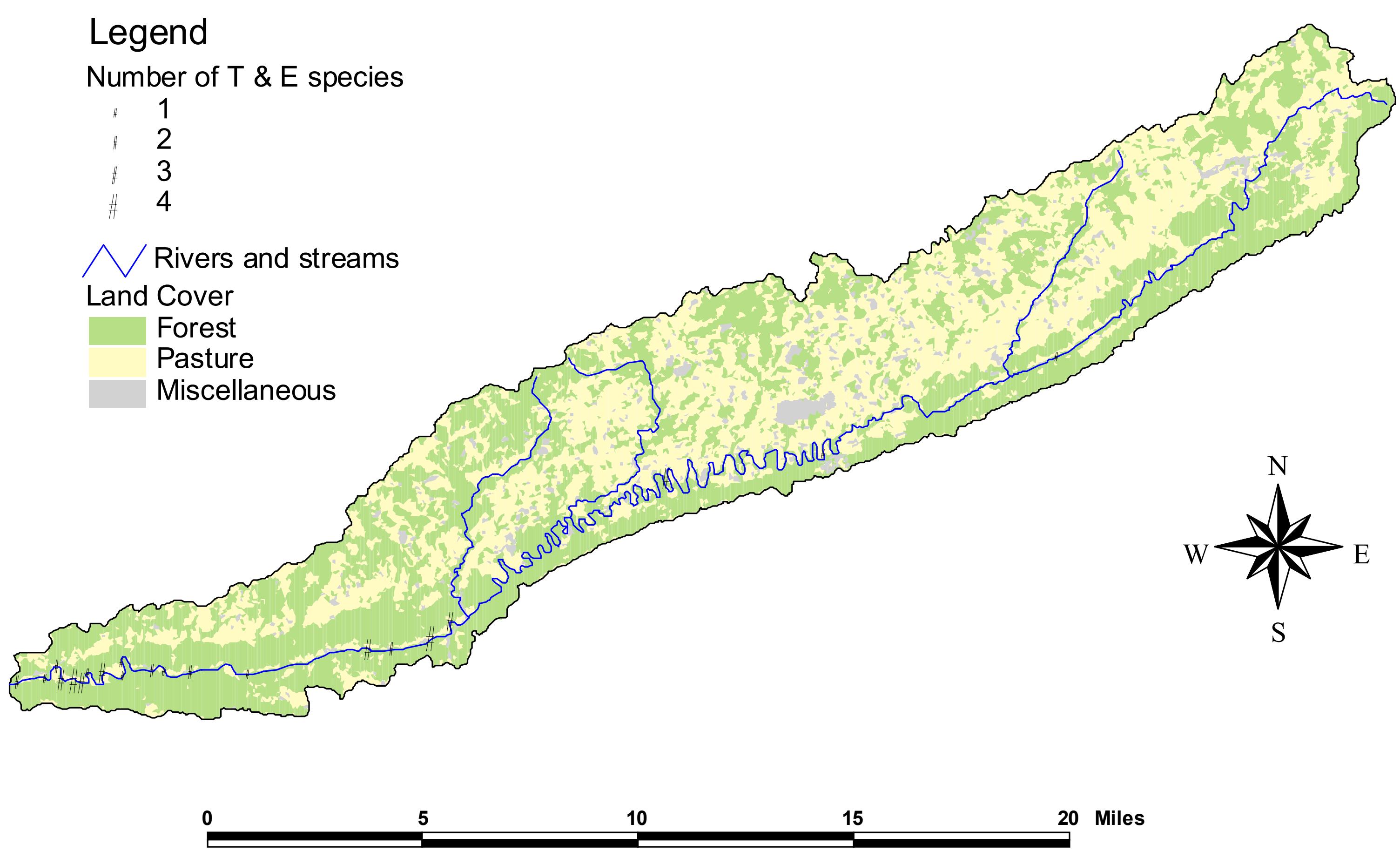
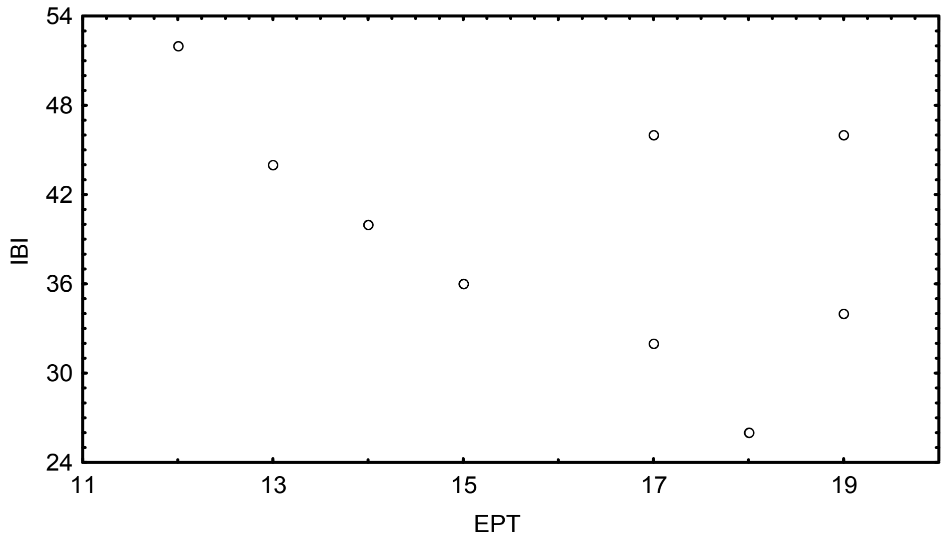


Figure 4-7. Number of threatened and endangered mussel species collected in different places on Copper Creek. Based on TVA's CMCP survey, 1981–1986.



Relationship between IBI and EPT for the entire CPRATS dataset

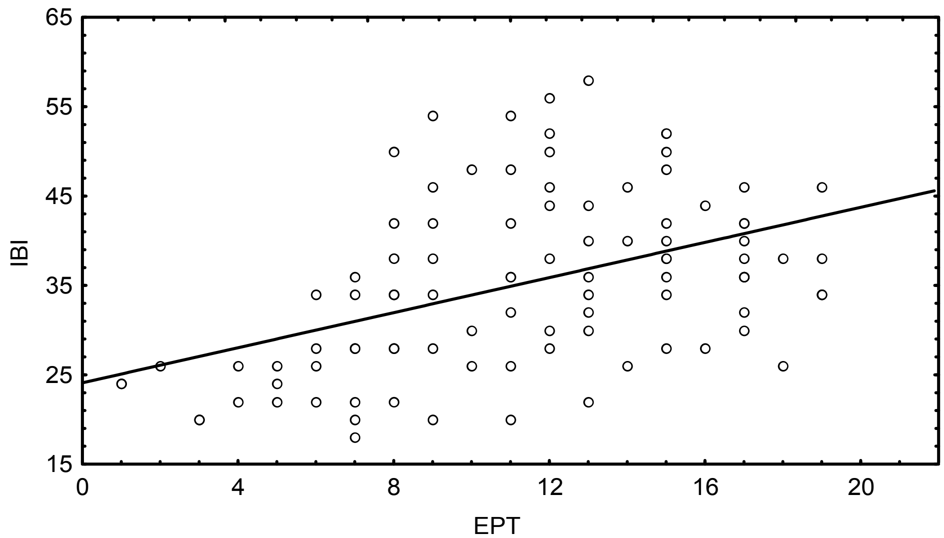


Figure 4-8. Relationships between benthic insect EPT and fish IBI values for TVA CPRATS sites (1995–96) in Copper Creek alone and for the entire Clinch and Powell CPRATS dataset.

[a]N = 9; r = 0.50
[b]N = 95; r = 0.52

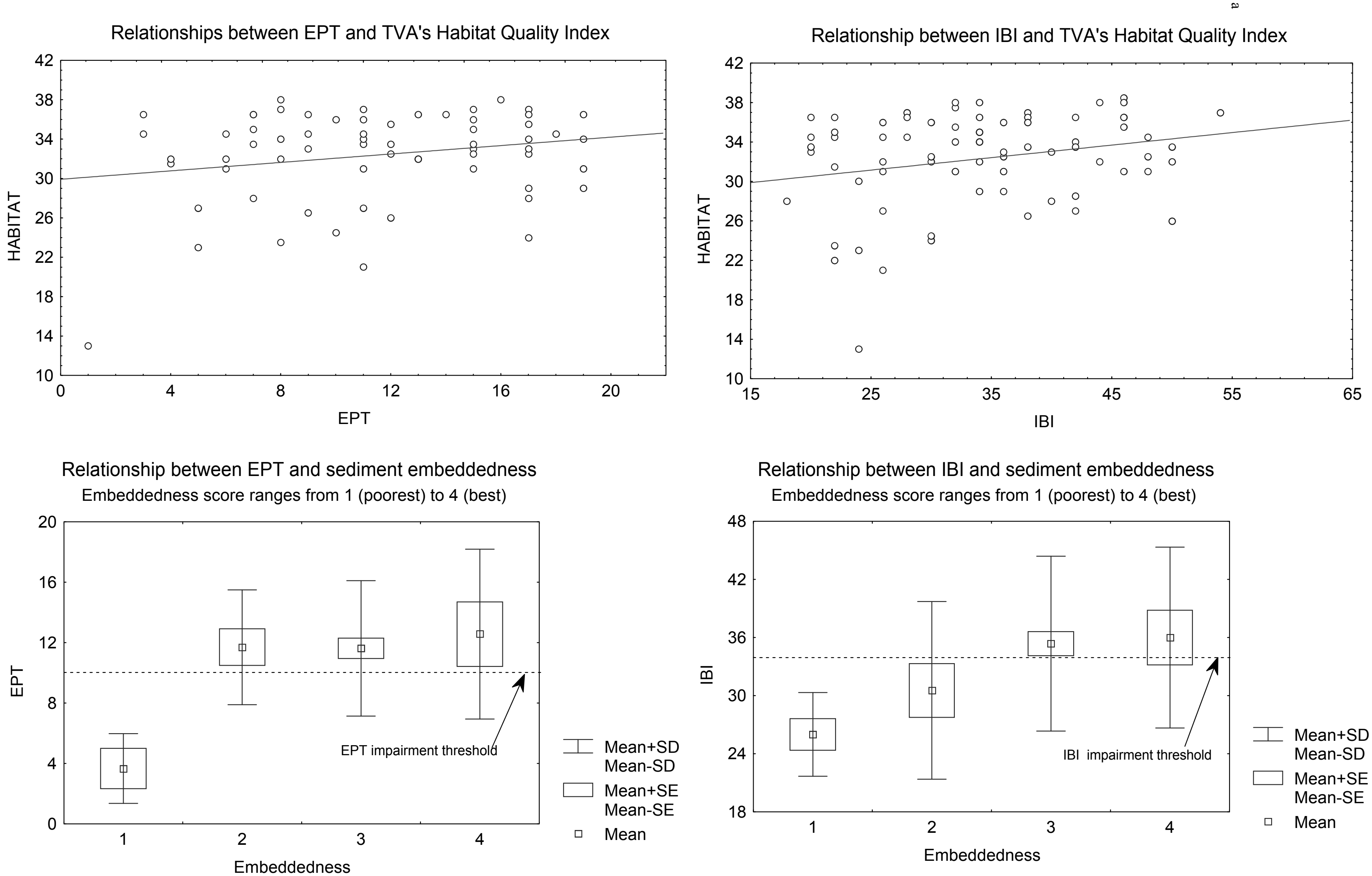


Figure 4-9. Relationships between TVA's multimetric stream habitat quality index or stream embeddedness score (TVA's scoring system) and either the invertebrate EPT (r = 0.22) or the fish IBI (r = 0.20) score for the Clinch and Powell CPRATS dataset (N = 95).

4.2.3. Analyses of Mussel and Fish Data in Copper Creek

We examined spatial relationships of native mussel and fish species richness in Copper Creek (Figures 4-1 and 4-6). Native mussel and threatened and endangered species data were derived from TVA's CMCP survey conducted in 1981 (32 sites). Fish data (35 sites) were obtained from Masnik (1974). Figure 4-10 plots the species richness values observed for both fauna as a function of stream river mile. With two exceptions (river miles 6–9 and 25–30) there was a reasonable fit between the two fauna because both generally tend to peak or trough at similar locations along Copper Creek. This suggests that both species richness measures were responding to similar stressors and/or that mussel species richness is in part dependent on the presence of fish host species. In general, greatest mussel species richness was observed in the lower 3 miles of Copper Creek, coincident with the large forested riparian area there. Areas in the "gorge," approximately 7 to 10 miles upstream of the mouth, had fewer mussel species, coincident with the fact that this area is within and directly downstream of a large agricultural (pasture) area of the watershed. However, this area had relatively high fish species richness.

Although these results could also be explained by a general increase in species with increased drainage area, fairly high fish and mussel species richness values were observed in the upper part of the creek, where upstream forested riparian area was also relatively extensive, though not as continuous as in the lower part of the creek. Overlaying mussel species richness values with the forest connectivity data layer (Figure 4-7) also suggested that mussels were responding, at least in part, to the extent of naturally vegetated riparian corridor area bordering Copper Creek. These results suggest that the extent of upstream riparian forested area is a critical factor that affects both mussel and fish assemblages in Copper Creek and that both types of fauna are not responding only to stream flow or drainage area.

Follow-up mussel sampling in Copper Creek in 1998 (Ahlstedt, 1999) indicated a general decline in the number of mussel species, the number of endangered species, and mussel abundance for most Copper Creek sites as compared with the 1981 CMCP survey (Figure 4-11). The report noted increased sedimentation at several sites, primarily because of livestock watering and runoff from pastures within the riparian corridor. These results indicate that riparian corridor integrity alone will not ensure adequate aquatic life habitat and maintenance of mussel populations if upstream sedimentation is severe enough.

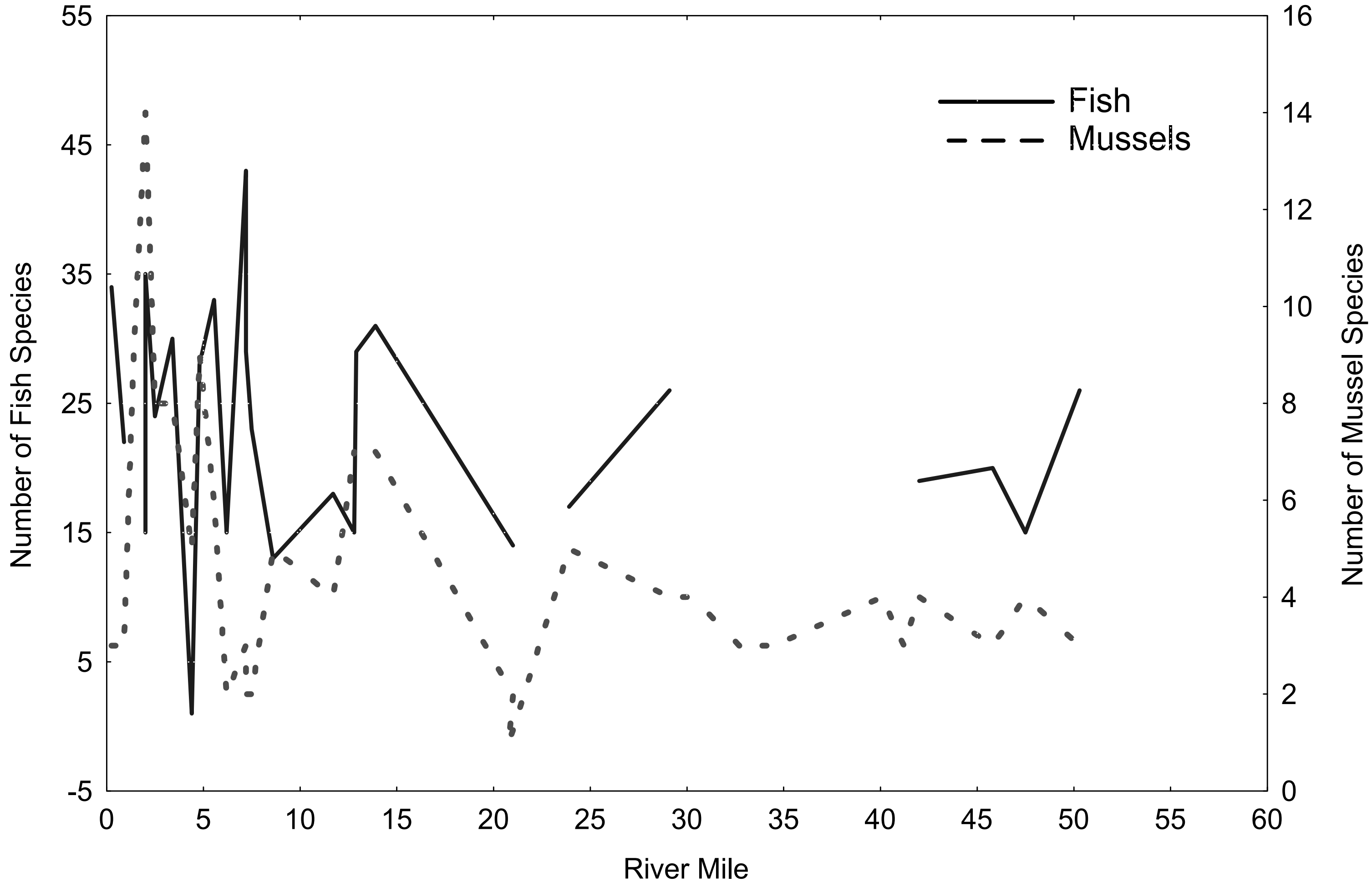


Figure 4-10. Fish and mussel species richness by river mile in Copper Creek. Fish data based on Masnik (1974); mussel data based on TVA CMCP (1981).

Mussel and fish data from Copper Creek support the hypothesis that both types of fauna are more responsive to riparian conditions than to more distant upland uses. Clearly, local stream geomorphic features also are likely to affect fish and mussel assemblages, because forested riparian corridor area only partially explained distribution and abundance of either fauna. Unfortunately, quantitative habitat quality measures were not collected concurrently with the fish or mussel data, and therefore we could not analyze habitat-biota relationships with those data. However, numerous studies have documented the important effects of local geomorphology and habitat quality on mussel (Stansbery et al., 1986; Church, 1996) and fish species richness (Angermeier and Smogor, 1993) in this watershed.

4.2.4. Temporal Comparison of Fish and Mussel Data in Copper Creek and Evaluation of Agricultural BMPs

In this analysis, we compared the fish species richness values collected by Masnik (1974) with the fish IBI values measured in CPRATS during 1995–96 and the mussel data collected by Ahlstedt in 1981 and again in 1998 (Ahlstedt, 1999). In the intervening years, agricultural BMPs designed to reduce sediment runoff into the stream were implemented at several sites in the Copper Creek watershed.

Figure 4-12 depicts fish results for those sites in which there were overlapping data from the two surveys. Data are expressed on a relative scale because we compared two different indices: the IBI and fish species richness. In each case, data for each site were expressed as a fraction of the highest value observed for the eight or nine sites available. The results suggested a similar trend with river miles in both time periods, possibly indicating similar conditions for the two time periods. Relatively higher fish IBI scores were observed in 1995–96 at river miles 2, 10, 12 and at the first of two sites at mile 43 as compared with other sites. This indicates relatively greater improvement in conditions at these sites in 1995–96 as compared with other sites. This result suggests some beneficial effect of BMPs, particularly at river mile 2, which was downstream of several BMP sites in the watershed. However, the mussel data shown in Figure 4-10, which were also collected before and after BMPs were implemented in Copper Creek, suggested little beneficial effect of BMPs on mussel populations. Indeed, Ahlstedt (1999) reported increased sedimentation at many sites, including several in the lower part of Copper Creek downstream of several BMP sites. Thus, contrary to the results for fish, it appears that the implemented BMPs had little if any beneficial effect on mussels and that sedimentation was not controlled. However, the extent and location of the BMPs, as well as the way in which they were performed in this subwatershed, may not have been sufficient to reduce agricultural sediment effects in Copper Creek..

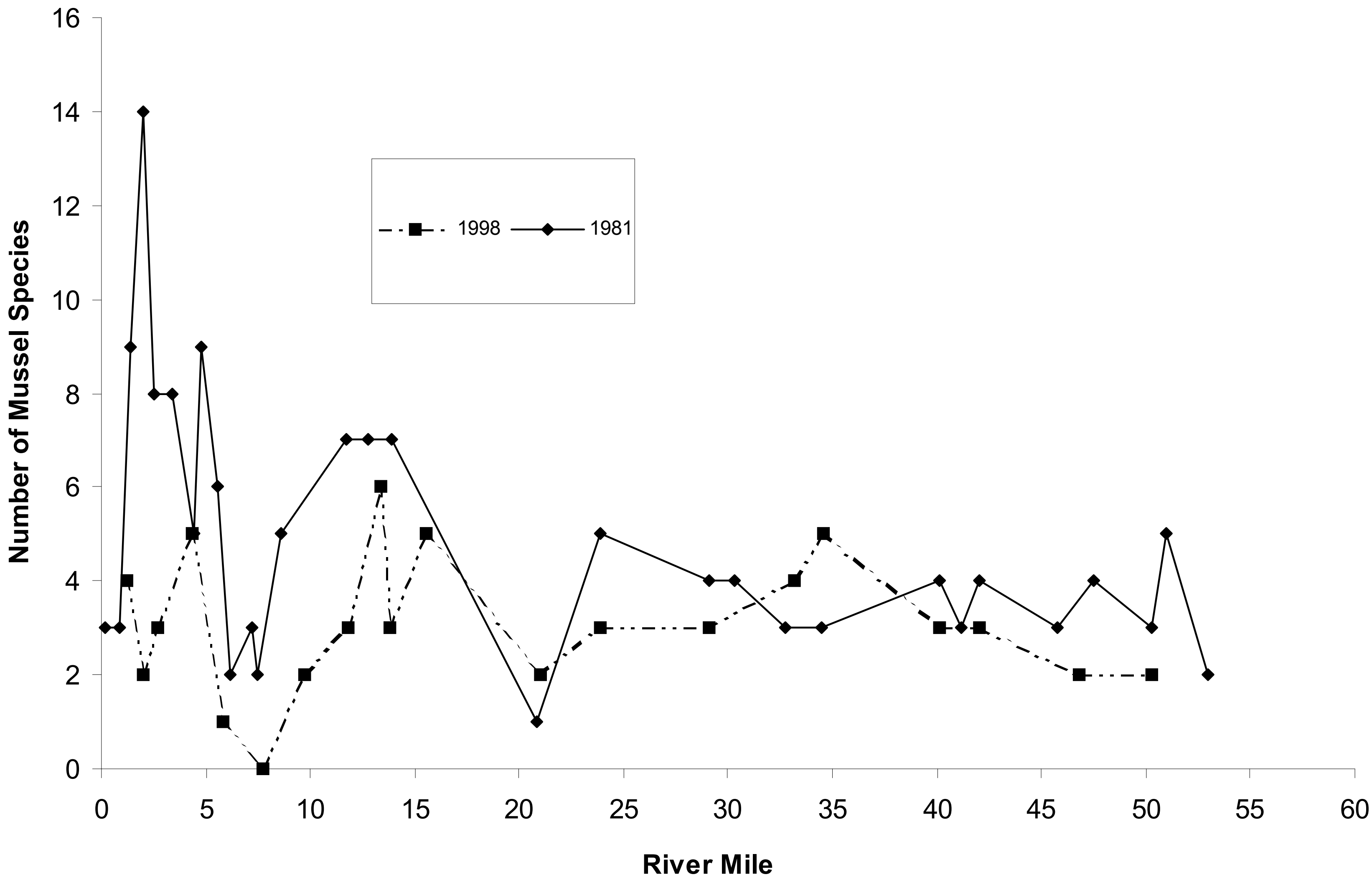


Figure 4-11. Number of mussel species in Copper Creek 1981 and 1998.

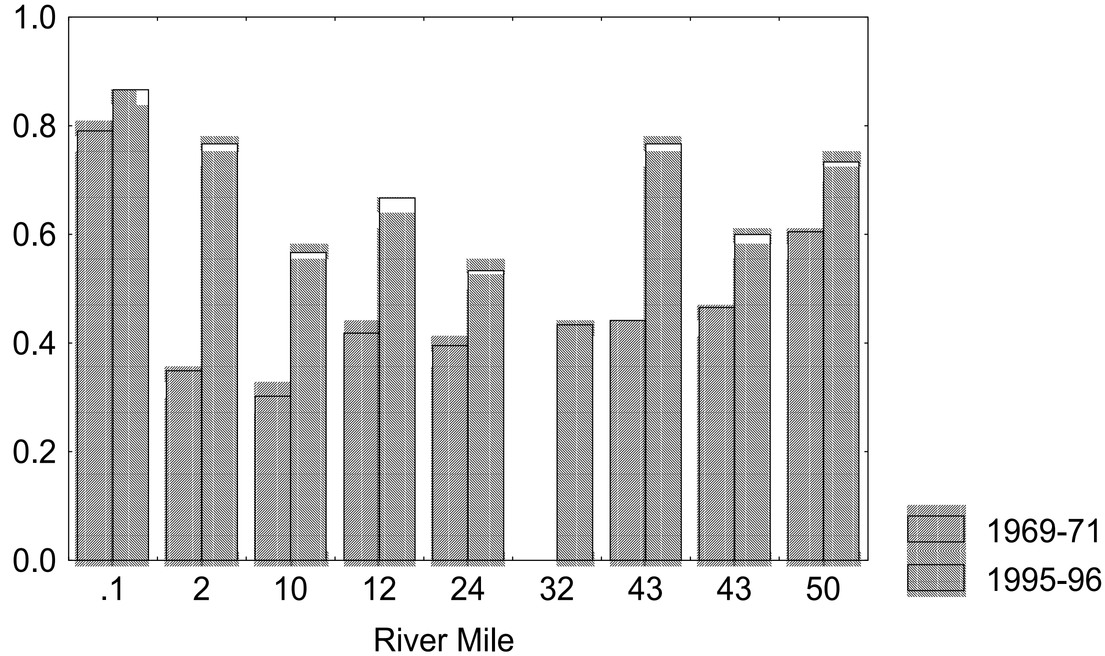


Figure 4-12. Fish IBI in 1995-96 compared with fish species richness observed in 1969–71 in Copper Creek. Data for each year area expressed as a relative fraction of the highest value obtained in that year.

4.3. RISK CHARACTERIZATION FOR COPPER CREEK PILOT TESTING AND FINDINGS FOR IMPLEMENTING WATERSHED ANALYSIS

The pilot testing yielded several important pieces of information, both for characterizing risks to Copper Creek and for understanding how to best conduct risk analyses for the watershed as a whole. The following results were identified:

- Intermediate spatial scales on the order of hundreds of meters appeared to be most relevant for relating land-use activities and biological and habitat measurement endpoints. Very small spatial scales (tens of meters) did not appear to adequately incorporate stressor effects on biological measures in a given site, whereas large scales (tens of kilometers) probably swamped out important sources of variation in biological attributes.

- A spatial scale on the order of hundreds of meters (100 m on either side of the stream by 500 to 1500 m upstream) appeared to be the most relevant for relating riparian corridor integrity and biological attributes. Furthermore, some measure of forested riparian connectivity also was useful in predicting biological attributes.

- The riparian corridor, as defined above, appeared to be more important than total upland land-use area in predicting native fish or mussel species richness. A riparian corridor composed of < 50% forest or < 500 m of continuous forest anywhere within a 1275 m segment had > 75% probability of being associated with impaired fish community integrity.

- Embeddedness and sedimentation were statistically significant habitat quality measures that affected the abundance and distribution of invertebrate and fish species in Copper Creek, and both habitat characteristics were directly related to the amount of agricultural land use in the riparian corridor.

- Fish taxa richness, measured around the time that mussels were sampled, was correlated with mussel species richness and suggested that fish community measures may be a useful surrogate endpoint for those areas in which mussel data are lacking. The EPT was not a reliable or sensitive measurement endpoint in Copper Creek, probably because this endpoint was based on family-level taxonomy, which is relatively coarse and dampens important expressions of variability in the invertebrate fauna. The IBI, on the other hand, was a more robust and sensitive indicator of human-induced perturbations in Copper Creek.

4.4. MANAGEMENT IMPLICATIONS

- Given limited resources, optimal benefits to fish, mussels, and perhaps other invertebrates would be realized by maintaining the riparian corridor for a minimum of 500–1500 m upstream and 100 m to either side of the stream for the site of interest. This riparian area could constitute a stream-specific optimal riparian management area within which to better prioritize protection efforts.

- Local riparian mitigation techniques (< 100 m upstream of the site) might not be as effective in enhancing fish or mussel diversity as somewhat larger riparian mitigation efforts. Local instream habitat characteristics may not be related to upland land uses if there is a wide vegetated riparian corridor in those areas.

5. RISK ANALYSIS FOR THE CLINCH AND POWELL VALLEY WATERSHED

Results of Copper Creek pilot analyses indicated that it was useful to analyze biological measures of effect, such as the fish IBI, and compare them with riparian corridor integrity, land use, and stream habitat quality measures. We extended these analyses to the CPRATS dataset as a whole, which comprised 153 sites located throughout the Clinch and Powell watershed. Data for fish, benthic macroinvertebrates, and habitat quality had been collected at many of these sites between 1995 and 1997.

In addition to using percentages of various major land uses (herbaceous-pasture, cropland, forest, and urban/barren land) as potential sources of stress, we also incorporated proximity to mining activities, major roads, and urban centers as sources. For these analyses, we created a coverage using ArcView, displaying the location of mines, coal preparation plants, or urban centers and CPRATS sites in the Clinch and Powell watershed. Using ArcView Spatial Analysis, 1- and 2-km buffers were created around each of the mines or urban centers. These distances were based on the results of the Copper Creek pilot testing, which suggested distances of 1 to 2 km upstream of a sampling point as being a relevant distance for evaluating land-use sources and effects. Three site categories were created, based on the location of the CPRATS sites: sites less than 1 km, sites 1 to 2 km, and sites greater than 2 km from a mine or urban center. Transitional sites that were on or very close to the 2-km border were eliminated. Proximity of a site to roads was included in this analysis as a surrogate indicator of sources of episodic spills. Transportation corridors, and particularly roads, have been sites of several truck accidents that resulted in spills of toxic materials. We also anticipated some habitat effects due to sedimentation from road construction and maintenance. Three classifications of roads were examined: four-lane State or U.S. paved roads, two-lane county or State paved roads, and dirt/gravel roads. The data were then aggregated into a single table and imported into Statistica for statistical analysis.

Forward stepwise multiple regression analysis was used to relate potential sources of stress (land-use percentages, proximity to urban or mining influences, and proximity to the three classes of road) to the fish IBI and the macroinvertebrate EPT. However, prior to examining effects of sources, we characterized effects of site elevation on biological measures of effect, because the entire CPRATS dataset covered a wide range of elevations and drainage areas, which could confound interpretations of land-use effects.

5.1. RELATIONSHIP BETWEEN STREAM ELEVATION AND BIOLOGICAL MEASURES OF EFFECT

The IBI exhibited a significant increase as elevation decreased (r = 0.54, $p<0.01$) (Figure 5-1), but the EPT and habitat quality measures were less related to elevation (r = –0.19, $p=0.05$ and r = –0.16, $p=0.16$, respectively). We expected greater fish and possibly EPT richness at lower elevations, because river reaches at lower elevations are broader and offer more diverse habitats for aquatic fauna (Vannote et al., 1980). Also, the available species pool or species richness is usually related to watershed area and, consequently, inversely related to elevation (Vannote et al., 1980; Karr and Chu, 1999). However, lower reaches also may be at greater risk from the cumulative impact of upstream stressors (Karr and Chu, 1999). Thus, competing factors may affect EPT species along a stream length, resulting in complex geographic patterns.

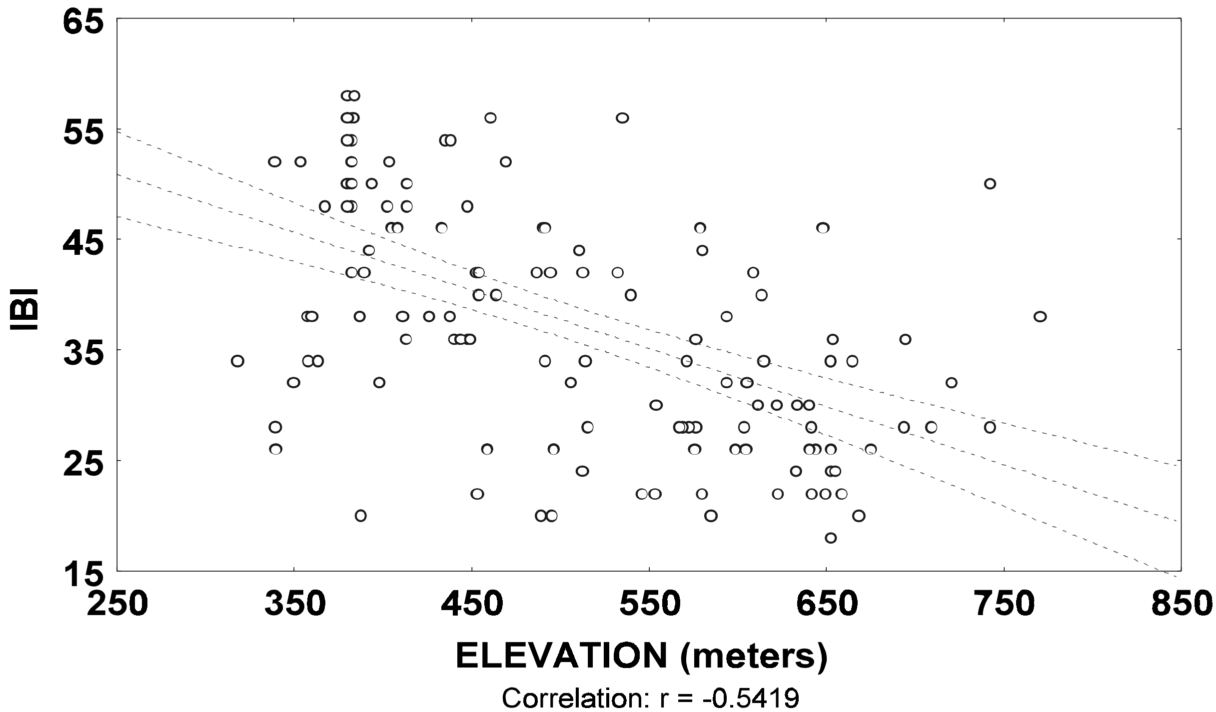
The effect of elevation on the IBI was best demonstrated by categorizing IBI scores as either "impaired" or "unimpaired" (using TVA's ratings) and comparing the mean IBI values for these two categories against elevation (Figure 5-1). t-Test analysis indicated significantly lower IBI scores at higher elevation sites ($p<0.01$). Cumulative frequency analysis indicated that sites higher than 500 m in elevation had better than an 85% probability of having unsatisfactory or impaired fish community integrity.

Because nearly half of the CPRATS sites that had concurrent habitat and biological information were located between 350 and 450 m elevation, and there was no elevation effect on the IBI or the EPT within this range, we concentrated subsequent multiple regression analyses for sites within 350 and 450 m elevation. A broader elevation range resulted in significant elevation effects. Fewer than 5% of the sites (5 out of 153) were < 350 m in elevation.

5.2. EFFECTS OF LAND USE ON HABITAT QUALITY MEASURES

Relationships between land cover and those stream habitat features measured by TVA in its habitat quality assessment (see Table 3-2) were analyzed using forward stepwise multiple regression analysis. This analysis produced few significant relationships between land uses or sources and habitat features reported at a given site, based on an elevation range of 350 to 450 m ($R^2 = 0.42$, $p=0.04$, N = 85). This result may be due to a lack of statistical power; we did not have a balanced representation of sites that had different land use combinations in our analyses.

Figure 5-2 summarizes significant relationships observed between land use and habitat. Stream sedimentation was lower where cropland was ≤3% of total land use. Riparian integrity was better in areas in which pasture/herbaceous land was < 50% of the total land use. Instream cover was poor if urban land use was ≥20% of the surrounding area upstream. Together, these relationships suggest that instream habitat will have the highest probability of being satisfactory



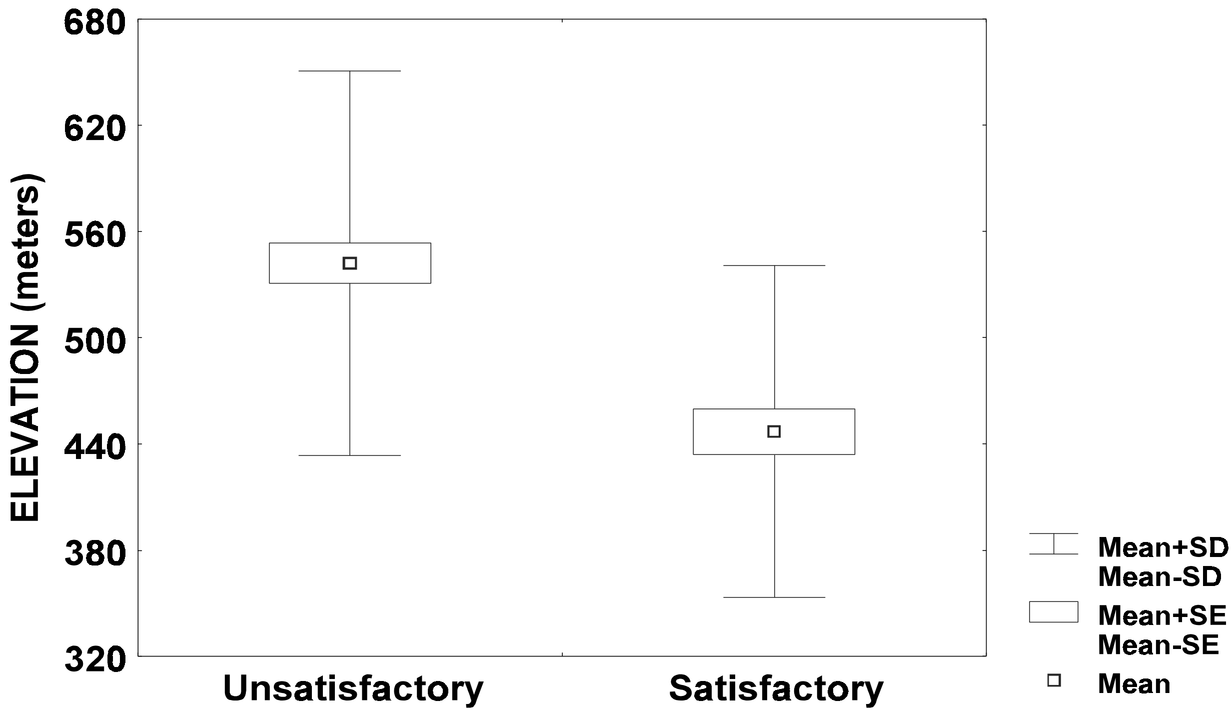


Figure 5-1. Relationship between the fish IBI and stream elevation, with the IBI expressed as either a continuous variable (upper figure) or as a categorical variable (lower figure), based on TVA's ratings.

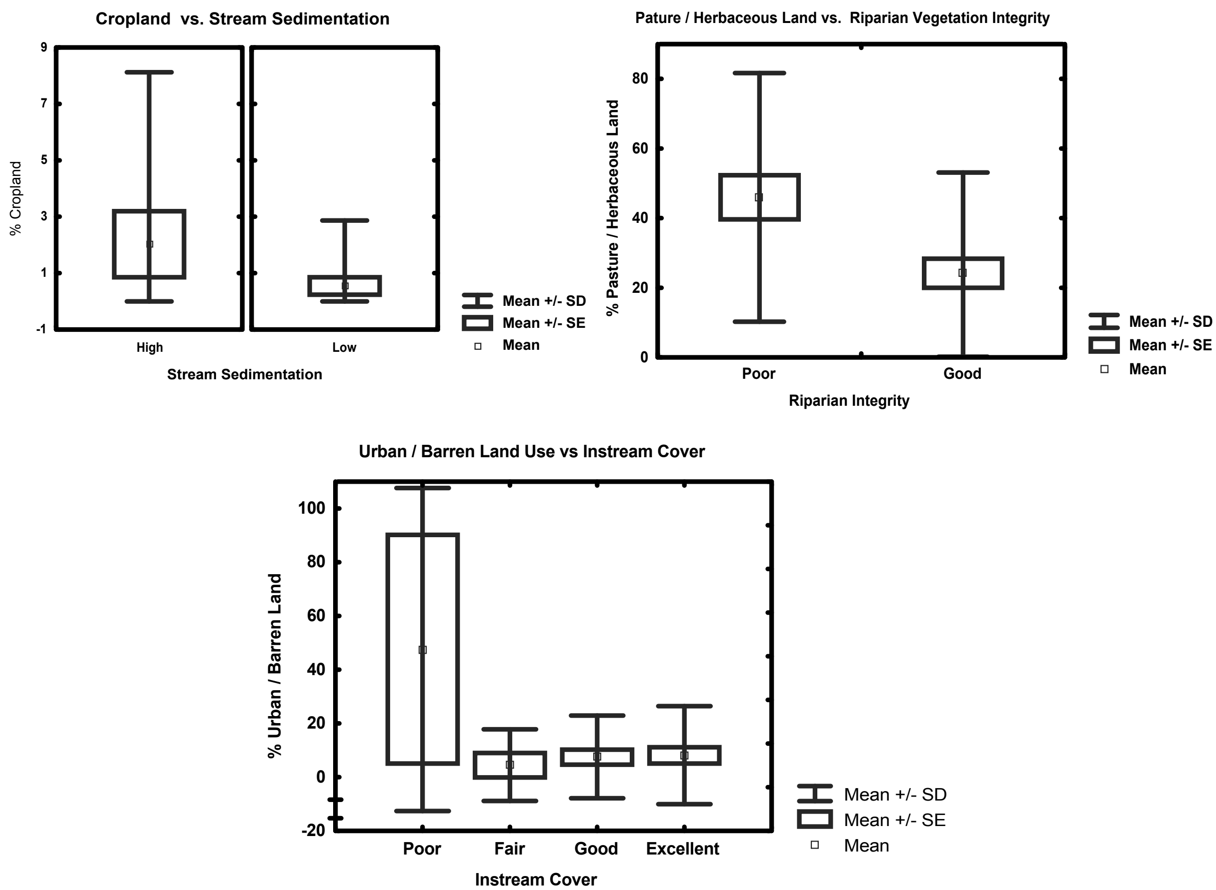


Figure 5-2. Relationships observed between land-use activities and instream habitat measures for sites in the Clinch and Powell watershed, based on TVA's CPRATS dataset and a restricted elevation range of 350–450 m (N = 85).

if cropland or pasture land is relatively low and urban influences are small or spatially removed. Proximity to roads did not appear to be significantly related to habitat measures (p>0.2).

Examination of sites at higher elevations (500 to 600 m) did not yield significant relationships between habitat measures and sources. However, some relationships were observed between stream channel stability and sources ($R^2 = 0.20$, p=0.08, N = 39). Sources in order of significance in the regression model were four-lane roads, percent cropland, proximity to active mining, and percent pastureland; however, p values for all source variables were > 0.13. No other habitat measures yielded a significant regression model. Thus, most of the variability in habitat measures of effect was not explained by the source data available.

Contrary to expectations, habitat quality scores and important habitat metrics, such as embeddedness or sedimentation, were not significantly different in relation to distance from coal mines for the entire CPRATS dataset (ANOVA, p>0.50). We anticipated an increase in fines and embeddedness at sites closer to active mines. These results may be due, in part, to the fact

that other land uses such as cropland or pasture also can result in increased sedimentation and, therefore, we were less apt to observe a specific effect of mining on sedimentation. Ample evidence from a number of sources has documented increased sedimentation from coal fines in the upper Powell and North Fork Powell rivers (Dennis, 1985; Wolcott and Neves, 1994; VDEQ, 1996), and these habitat impacts were highly correlated with decreases in mussel species richness and abundance.

Some interactions between land-use factors and resultant habitat features were evident, as shown in three-dimensional contour plots (Figure 5-3). Instream cover and embeddedness were affected by both the percent pasture/herbaceous cover and the percent urban area nearby. Both land uses contribute sediment to the stream and reduce available substrate diversity, which contributes to embeddedness and poor cover for fish and invertebrates. The lower the percentage of either use, the better the instream cover and the lower the instream embeddedness (Figure 5-3A and 3B, respectively). This result is consistent with the fact that instream cover is, in part, inversely related to the amount of embeddedness present. However, we observed unexpectedly high embeddedness and instream cover scores (i.e., good cover and low embeddedness) at intermediate urban percentages (~30% urban cover) when percent pasture cover was low
(< 30%). This result probably illustrates the fact that certain land-use combinations are less represented than others, which is a limitation of the available data set. This data constraint is a potential confounding factor in risk analysis for any watershed, given the general sparcity of available data.

Riparian corridor integrity was affected primarily by the percent pasture/herbaceous land upstream (as observed in Copper Creek), but mining or urban proximity also appeared to play a role: in both cases riparian integrity suffered with close proximity to mining or urban influences (Figure 5-3C and 3D, respectively). Thus, a combination of both agricultural area in the immediate vicinity of a site and urbanization or mining upstream appeared to yield poor riparian integrity.

5.3. RELATIONSHIPS BETWEEN LAND USE AND BIOLOGICAL MEASURES OF EFFECT

5.3.1. The Fish IBI

Given the previous established relationship between elevation and the fish IBI (Figure 5-1), these analyses focused on sites at an elevation between 350 and 450 m, for which elevation was not a confounding effect (*p*>0.2). Forward stepwise multiple regression analyses indicated that proximity to mining was the most significant factor related to the IBI values and that percent

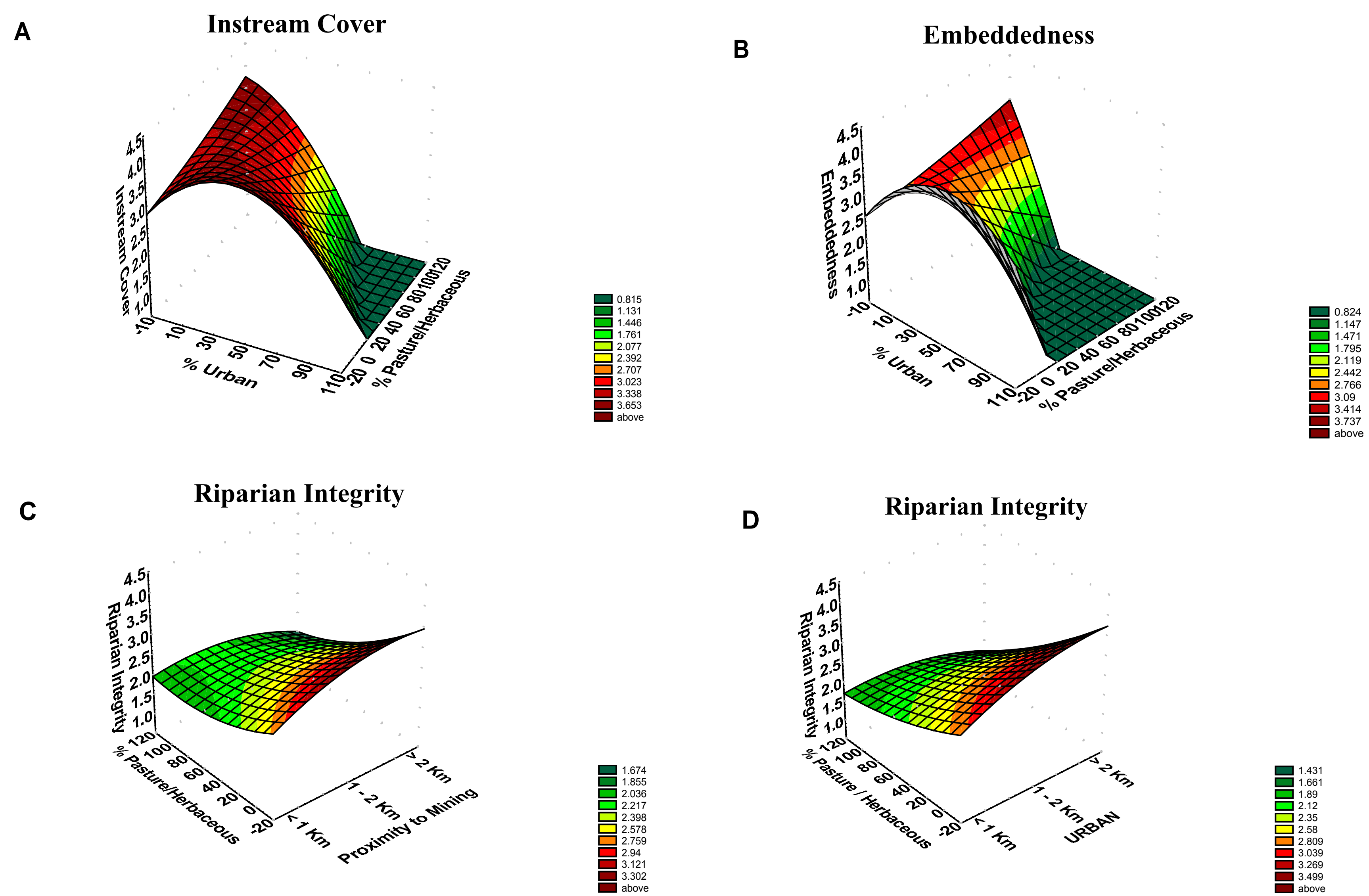


Figure 5-3. Three-dimensional contour plots illustrating two-way interactions observed between land uses and effects on instream habitat features. The Y-axis in all cases represents habitat quality metrics that reflect better condition with higher scores (e.g., a high embeddedness score means very little embeddedness observed instream). Data were from TVA's CPRATS dataset and an elevation range of 350–450 m (N = 85).

Table 5-1. Summary of forward stepwise multiple regression analyses of fish IBI values, obtained in TVA's CPRATS dataset for the Clinch and Powell Watershed, and land-use factors

Model	N	R^2	Influencing factors	Association	Partial R^2	*p*
CPRATS sites between 350 and 450 m in elevation	38	0.55	Proximity to mining	–	0.18	0.013
			Percent pasture area	+	0.15	0.019
			Percent cropland	–	0.06	0.15
			Percent urban	–	0.04	0.23

pasture area was directly related to the IBI, whereas proximity to mining, percent cropland, and percent urban land were inversely related (Table 5-1). Proximity to roads was not a significant factor in the regression model. However, 45% of the variability in IBI scores was unexplained by our model, indicating that (1) the composite nature of IBI scores may conceal relationships between fish assemblage and land uses; (2) other site-specific factors, such as hydrologic regime, proximity to accidental chemical spills, or other water quality effects are significant sources of stress in this system; or (3) statistical power was hampered by unequal representation of sites with different land-use combinations.

The relatively beneficial effect of pasture/herbaceous land on fish community integrity for the watershed as a whole and at mid-elevations (Figure 5-4) was unexpected, based on the pilot analyses using data from Copper Creek only (section 4.2). However, it should be noted that extrapolation between watersheds in these analyses was subject to some uncertainty because stream size is not constant across subwatersheds (but see section 5.6). The indication of a relative beneficial effect of pasture/herbaceous land on fish community integrity probably resulted because (1) there is historically greater livestock pressure in Copper Creek than in most other agricultural areas in the Clinch and Powell watershed (Don Gowan, TNC, at the Clinch assessment workgroup meeting, December 1, 1998), (2) pastures have potential nutrient enrichment effects on fish communities, and (3) mining and urban areas are comparatively far more detrimental sources of stress than pasture areas as a whole on fish in this watershed. In fact, we observed that percent forested land cover was greater near mining activity than it was farther away (ANOVA, N = 152, F = 5.93, p=0.003), and it was negatively correlated with pasture land cover (r = –0.80, p<0.05). As a result, sites with higher IBI scores (i.e., better fish community integrity) were associated with *less* forested cover than sites with lower IBI scores (poorer fish community integrity; N = 137, t = 3.01, p=0.003). Thus, this analysis confirms that proximity to mining has a profound effect on fish communities in this watershed.

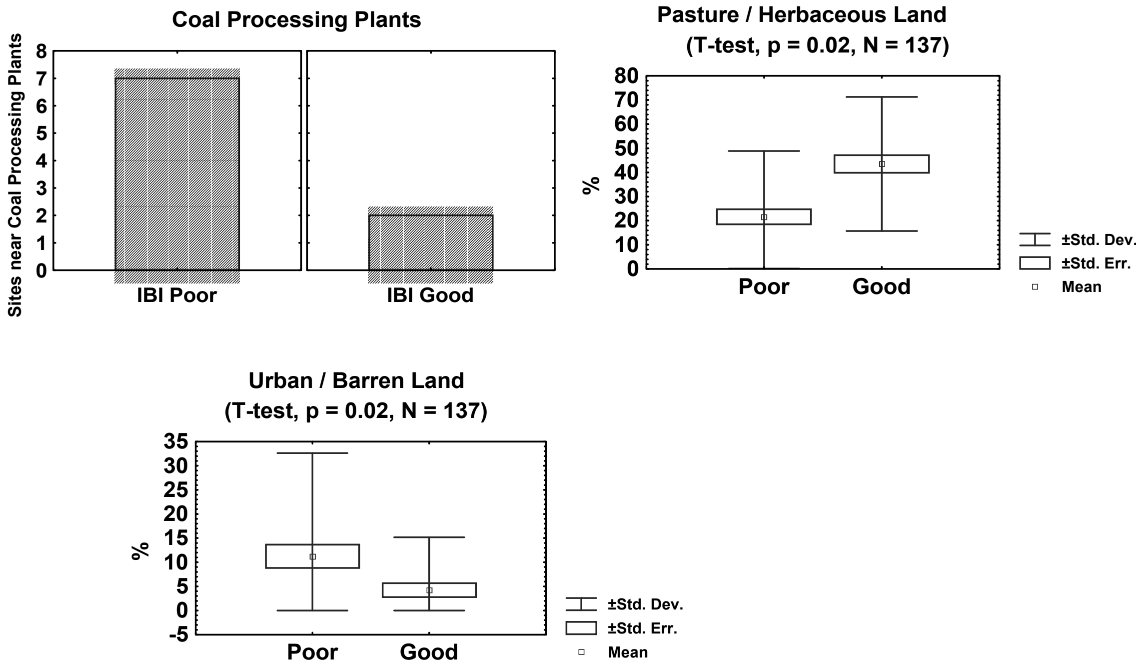


Figure 5-4. Significant relationships observed between land-use sources and the IBI for the entire CPRATS data set (N = 155).

Mining and urban areas appeared to have clear, negative effects on the IBI (ANOVA, $p<0.01$) (Figures 5-4 and 5-5). One potential stressor from mining is discharge of toxic contaminants. For example, a prior study (BMI, 1990) found that the hydraulic fluid typically used in longwall (deep) coal mine machinery is highly toxic to many types of aquatic species and especially to larval mussels (Figure 5-6). Although actual exposure concentrations of this material in the stream have not been documented, it is known that at least two different fish kills on the North Fork Powell (1986 and 1988) were caused by acutely toxic discharges of hydraulic fluid (BMI, 1990). Water-quality impacts from coal mining are also suggested by examining the relative effects of different types of mining activities on the fish IBI. Figure 5-7 shows that the fish IBI is more depressed downstream of either coal processing plants or surface mines, as compared to sites with no mining activity present (ANOVA, $p< 0.05$). Both of these coal mining activities contribute water quality and physical (coal fines) stresses on aquatic biota.

A recent study conducted by FWS (Lingenfelser, 2000) on samples collected from several coal processing plant discharges in the Clinch and Powell watershed yielded similar information. Chemical and toxicological quality of the samples as well as benthic macroinvertebrate assemblage integrity were evaluated by FWS downstream of four different facilities. Several species were subjected to toxicity testing, including glochidia and juveniles of two surrogate

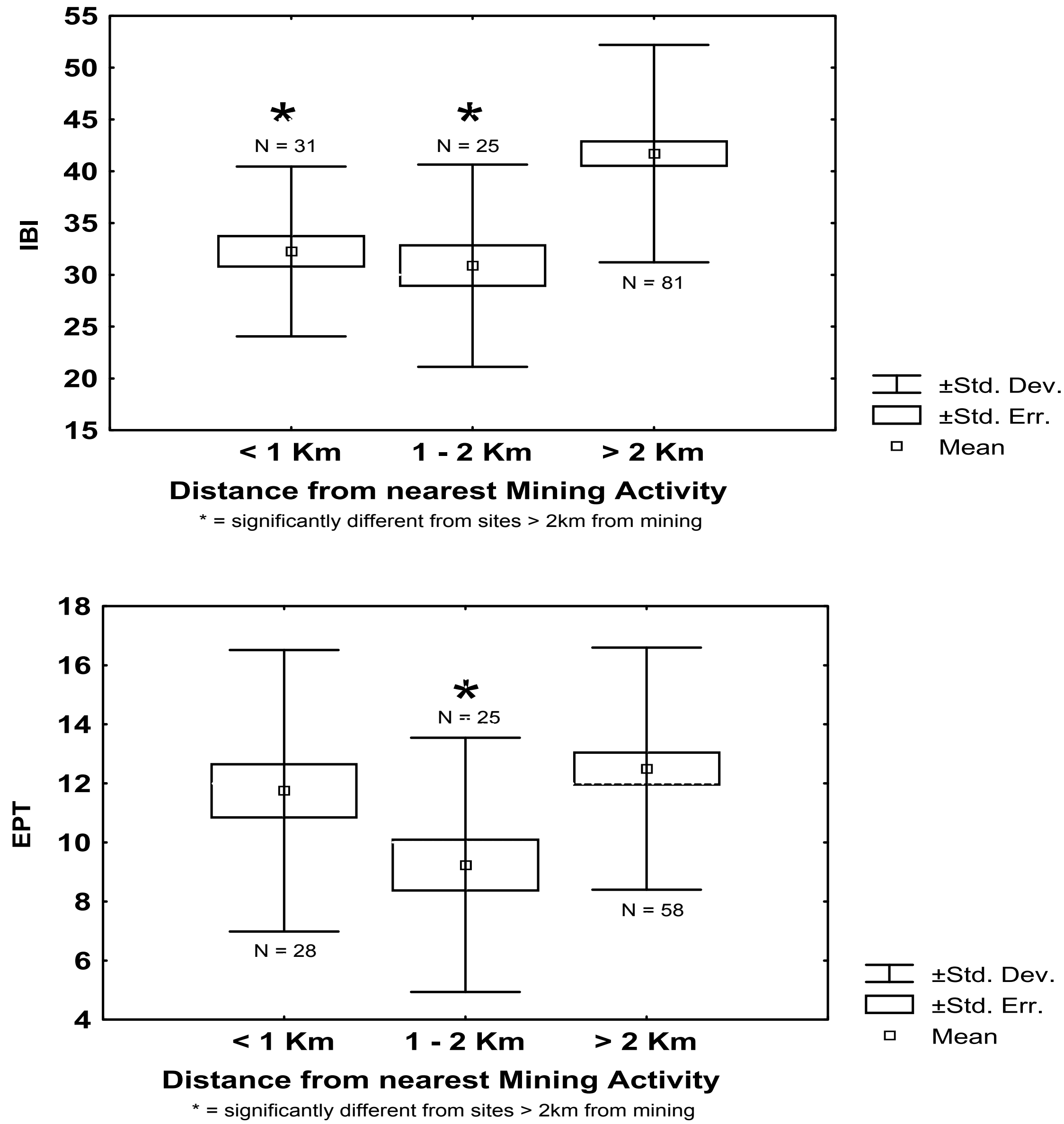


Figure 5-5. Fish IBI or insect EPT values in relation to proximity to coal mining sources. Data derived from TVA's CPRATS dataset (1995–1997).

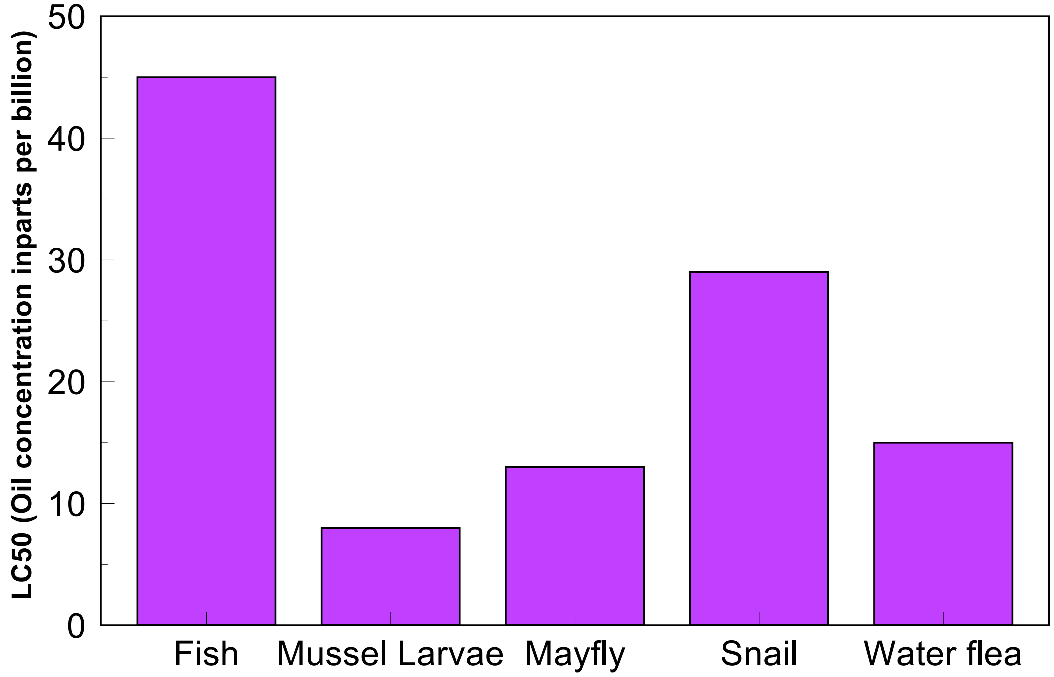


Figure 5-6. Comparison of acute toxicity results (LC_{50}) for several species, based on laboratory water exposures of a hydraulic oil commonly used in coal longwall mining machinery in the Powell River subwatershed. The higher the LC_{50}, the less toxic the chemical. Actual species tested included *Pimephales promelas* (fathead minnow), *Villosa spp.* (mussel), *Stenonema spp.* (mayfly), *Physella spp.* (snail), and *Ceriodaphnia dubia* (water flea).

Source: BMI 1990.

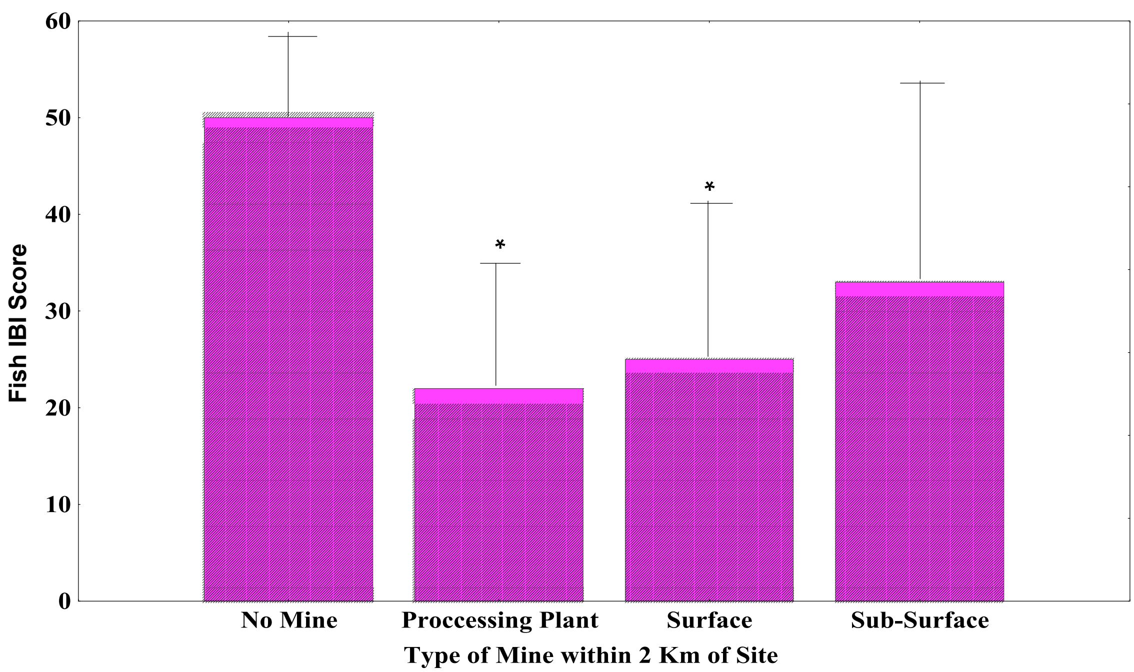


Figure 5-7. Effect of mine type on fish community integrity as a function of the type of mine.

* = significantly different from sites having no mining activity present within 2 km.

mussel species, the amphipod sediment indicator species *Hyalella azteca* and the EPA freshwater indicator species *Ceriodaphnia dubia* and *Pimephales promelas*.

Iron, nickel, and selenium occasionally exceeded EPA water quality criteria at the process discharges, and several other metals approached water quality criteria. No other contaminants were detected in these samples. A few samples were reportedly toxic to *Ceriodaphnia* and *Hyalella*; however, data were limited both in terms of number of samples and types of sites tested. Benthic macroinvertebrate sampling indicated generally poor assemblage integrity downstream of all coal processing discharge sites, as exemplified by fewer taxa, number of individuals, pollution-sensitive taxa such as the EPT, and lower diversity of taxa, as compared with either reference or upstream sites. Thus, these preliminary data also indicate deleterious effects of coal processing activities on aquatic life in the Clinch and Powell watershed.

The impacts of urbanization on watersheds are well documented (Jones and Clark, 1987; Karr, 1991; Schueler, 1994). They include increased sediment loads, nutrient input, and toxic input. These problems are exacerbated by the increase in impervious surfaces prevalent in urban areas. Stream degradation occurs at relatively low levels of imperviousness (10–20%) (Schueler, 1994).

One site-specific factor that could be important is point-source pollution such as industrial or municipal wastewater discharges. Effects of point sources were not explicitly included in our analyses because of the few significant dischargers and the fact that many of the major ones were assumed to be encompassed in the urban land use classification. Other information collected during this risk assessment indicated significant effects of large point sources on native mussels and other aquatic life (Cairns et al., 1971; Crossman et al., 1973; Goudreau et al., 1993).

Another stressor difficult to characterize in this risk assessment is catastrophic spills of toxic materials. Figure 5-8 summarizes the types of effects observed after catastrophic spills at Westmoreland Coal Company and the APCO power plant on the Powell and Clinch rivers, respectively. In 1998, a large coal slurry impoundment on the upper Powell River failed, resulting in a massive fish kill and substantial mortality of native mussels for a distance of more than 20 miles downstream (Hylton, 1998). A 1999 truck accident on the upper Clinch River in the Cedar Creek area resulted in substantial loss of mussels, including more than 300 threatened and endangered mussels.

5.3.2. The Macroinvertebrate EPT

The invertebrate EPT score exhibited more limited relationships with sources than did the IBI. Forward stepwise multiple regression analyses indicated that the EPT was related to percent

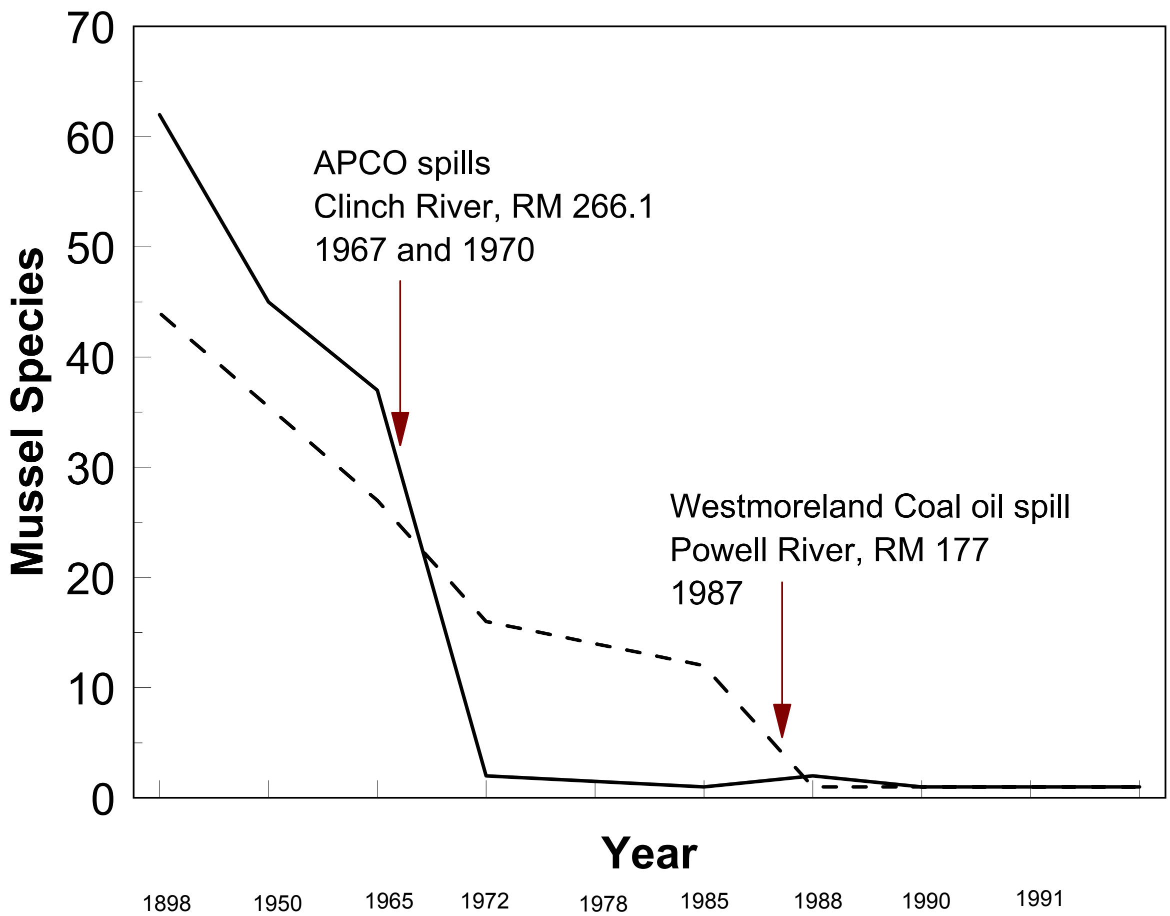


Figure 5-8. Number of mussel species recorded over time at two sites in the Clinch and Powell watershed affected by large toxic point-source discharge events.

urban area, proximity to roads, and percent pastureland for sites between 350 and 450 m in elevation, but the overall R^2 was much lower than that for the IBI ($R^2 = 0.29$) (Table 5-2). Percent urban area nearby showed the clearest effect on the EPT, particularly when examined using categorical data (Figure 5-9). Proximity to mining, which had the most impact on the IBI, exhibited less of a relationship with the EPT (Figure 5-5).

As we indicated previously, the less significant relationships between the EPT and land cover could be due to the coarser taxonomy used for invertebrates and the resultant loss of information. However, we cannot rule out the fact that invertebrates have shorter life cycles than do fish, and they may be able to recolonize or recruit individuals more quickly following stress than can fish. Several researchers have noted relatively rapid recolonization of macroinvertebrates following episodic events such as chemical spills (Cairns et al., 1971; Crossman et al., 1973), pesticide applications (Wallace et al., 1986), and physical disturbances such as severe floods (Fisher et al., 1982; Minshall et al., 1983). Site-specific data collected before and after the APCO chemical spill in the Clinch River in 1970 demonstrated that

Table 5-2. Summary of stepwise multiple regression analyses of macroinvertebrate EPT value in relation to potential sources of stressors in the Clinch and Powell watershed. Based on TVA's CPRATS data set.

Model	N	R^2	Influencing Factors	Association	Partial R^2	*p*
EPT sites 350–500 m elevation	34	0.29	Percent urban area	–	0.14	0.033
			Percent pasture	+	0.08	0.127
			Proximity to mining	–	0.07	0.145
			Proximity to roads	–	0.04	0.310

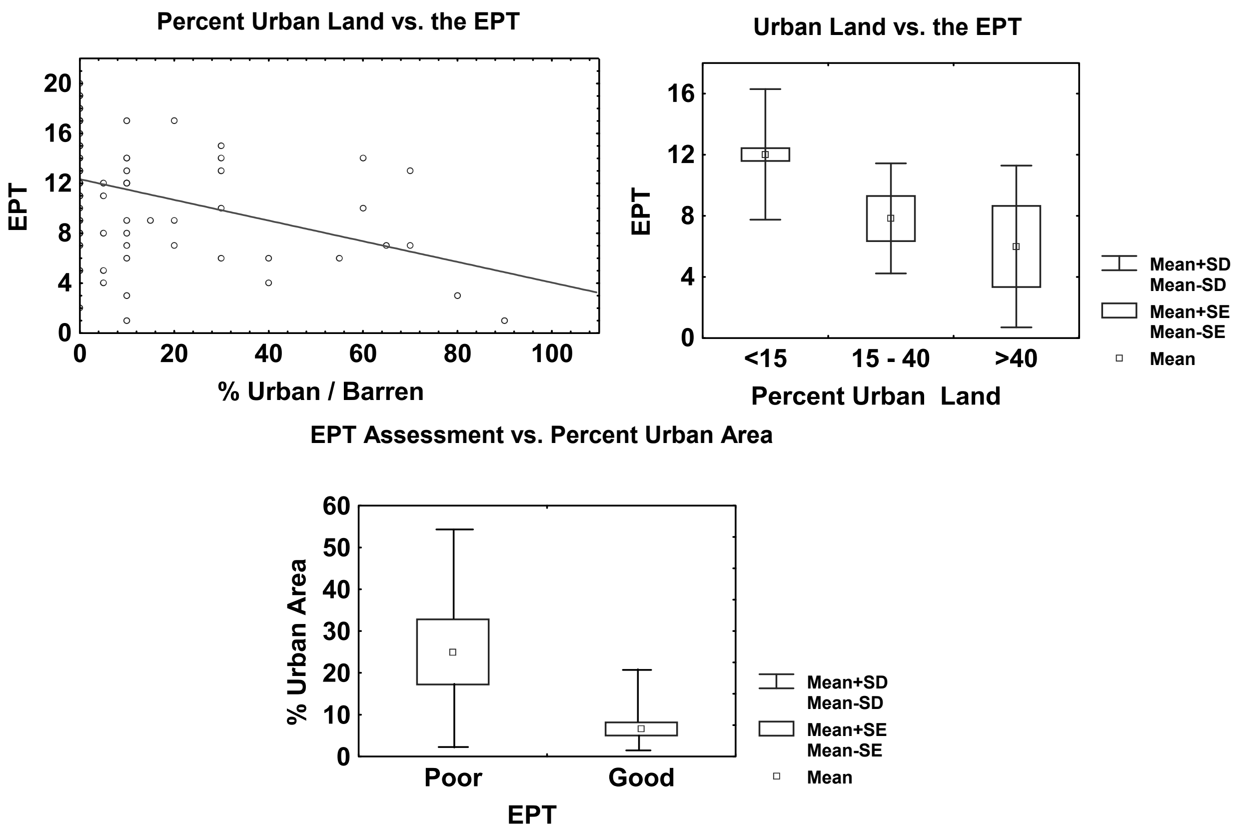


Figure 5-9. Significant relationships between land-use activities or habitat quality and invertebrate EPT score. ($R^2 = 0.29$)

macroinvertebrates recolonized damaged areas relatively quickly (within 1 year) (Crossman et al., 1973), whereas native mussels and fish have largely not recolonized this area of stream as of this writing (Figure 5-8). Thus, the EPT may recover from spills or other episodic events relatively quickly and, therefore, may not be as sensitive an indicator of past water quality effects as either native mussels or fish.

5.3.3. Mussels

TVA's CMCP data were used for these analyses. Unfortunately, data for only 33 sites (in addition to the 32 sites in Copper Creek discussed in section 4.2) were readily available, although far more data are known to be archived in TVA's database. Resource constraints did not allow these other data to be retrieved and incorporated into the GIS. Other published information was used, when relevant, to help supplement our analyses and aid in interpreting the data.

CMCP data were available for river miles 73 to 166 on the Powell River and 159 to 322 on the Clinch River. Data available for each site included number of mussels, number of native and endangered species collectively, and number of endangered species only.

Forward stepwise multiple regression analysis indicated that none of the riparian land use factors were significantly related to mussel density for both rivers combined or for either river separately ($p>0.20$). A significant model could not be constructed for native mussel abundance, indicating that the factors available had little explanatory value for this measure of effect. Number of native mussel species, however, was related to several variables, including (in order of significance) percent urban area, proximity to mining, and percent cropland (Table 5-3). Sites further upstream, far from towns or mining, or having agricultural land use instead of urban or mining activity directly around and upstream of the site tended to have a greater number of mussel species present ($R^2 = 0.26$, $F = 3.01$, $p<0.03$).

However, this model explained only 26% of the variability in mussel species richness, indicating that other unmeasured factors affect mussel distribution in these rivers. As most of these sites were located on the mainstem rivers, land-use combinations were limited in our analysis, reducing statistical power. The likely factors contributing to the unexplained variability in mussel species richness are (1) site-specific geomorphic characteristics such as substrate particle size, flow and current velocity, and orientation of bedrock ridges (Church, 1991), (2) proximity to episodic spills that could not be adequately analyzed in this risk assessment, and (3) fish host assemblage in the area varies from year to year.

Subsequent analyses of mussel data in the upper Clinch subwatershed indicated significant relationships between riparian land-use factors and mussel species richness,

particularly if geomorphological factors such as drainage slope are taken into account (see section 5.6).

Table 5-3. Summary of forward stepwise multiple regression analysis of Cumberlandian mussel species richness as a function of riparian land-use factors. Data are from TVA's CMCP database

Model	N	R^2	Significant factors	Association	Partial R^2	*p*
All CMCP sites	33	0.26	Percent urban area	–	0.09	0.007
			Proximity to mining	–	0.05	0.05
			Percent cropland	–	0.02	0.30

Results of the foregoing analyses in this section indicate that riparian land uses accounted for, at most, 55% of the variability in biological measures, although certain land use–response relationships were clearly evident. The riparian corridor dimensions used in our analyses were derived from data for the Copper Creek subwatershed and then extrapolated to other streams. However, it should be noted that the assumptions about extrapolating from small to large systems must include system characteristics (see section 5.6). Larger streams such as the upper Powell or upper Clinch rivers could conceivably have different relationships between land uses and biological measures. Thus, upland land uses or larger riparian areas may need to be considered in future assessments for this watershed to confirm relationships between land uses and biological measures of effect.

5.4. RELATIONSHIPS BETWEEN HABITAT MEASURES AND BIOLOGICAL MEASURES OF EFFECT

Table 5-4 summarizes results of stepwise multiple regression analyses on habitat measures and both the fish IBI score and the macroinvertebrate EPT. These analyses used the entire elevation range of sites in the CPRATS dataset because elevation was uncorrelated with habitat measures ($p>0.2$). In addition to the individual habitat metrics measured by TVA, the overall habitat score for a site reported by TVA was also used as an independent variable in the analysis because it was uncorrelated with any of the metrics ($R = 0.20$, $p>0.10$).

Figure 5-10 shows relationships between either embeddedness (or the inverse, clean sediment in the figure) or instream cover and the fish IBI, categorized as either poor or good (based on TVA's criteria). In both cases, better IBI scores were associated with more cover and clean substrate. Streams with either embeddedness scores of ≤ 2.0 or instream cover scores of

< 3 had a greater than 90% chance of having poor fish community integrity. Given that we observed negative relationships between pasture/herbaceous land cover or urban proximity and embeddedness (Figure 5-2), it is not surprising that both of these land-use activities have effects

Table 5-4. Summary of forward stepwise multiple regression analyses of habitat quality measures in relation to either the fish IBI or the macroinvertebrate EPT. Data are from TVA's CPRATS dataset for sites between 350 and 500 m in elevation

Model	N	R^2	Significant Factors	Association	Partial R^2	*p*
IBI	81	0.29	Instream cover	+	0.12	0.002
			Channel stability	+	0.10	0.006
			Embeddedness	–	0.07	0.024
			Habitat score	+	0.06	0.026
			Epifaunal substrate	–	0.01	0.252
EPT	65	0.23	Channel stability	+	0.10	0.014
			Epifaunal substrate	–	0.06	0.055
			Instream cover	+	0.04	0.105
			Habitat score	+	0.05	0.067
			Embeddedness	–	0.02	0.228

on fish assemblage integrity (Figure 5-4). The EPT was significantly related to instream channel stability and epifaunal substrate, a measure of the substrate complexity or heterogeneity in particle size and woody snag material for benthos (Figure 5-11). Both of these habitat measures are important features that directly affect macroinvertebrate diversity (Karr and Chu, 1999; Barbour et al., 1999).

The relatively small variability in biological measures explained by habitat measures in these analyses (total variance explained in either the EPT or the IBI does not exceed 30% in regression analyses) (Table 5-4) suggests that chemical stressors may play a significant role. Unfortunately, we were unable to characterize chemical stressors because of a lack of relevant data. There are only two long-term water quality stations in the entire watershed, both located in the lower part of the watershed. Potential chemicals of concern, such as pesticides, coal mining chemicals, and heavy metals were largely unmeasured. Thus, water quality stressors were inferred in this risk assessment on the basis of nearby land use/source activities in association with biological effects and habitat quality information. However, the habitat data used were highly qualitative, which may add uncertainty to these analyses. Our experience suggests that some investigator bias cannot be avoided in using such qualitative assessment protocols. These results underscore the difficulties in deriving stressor-response relationships on a watershed scale.

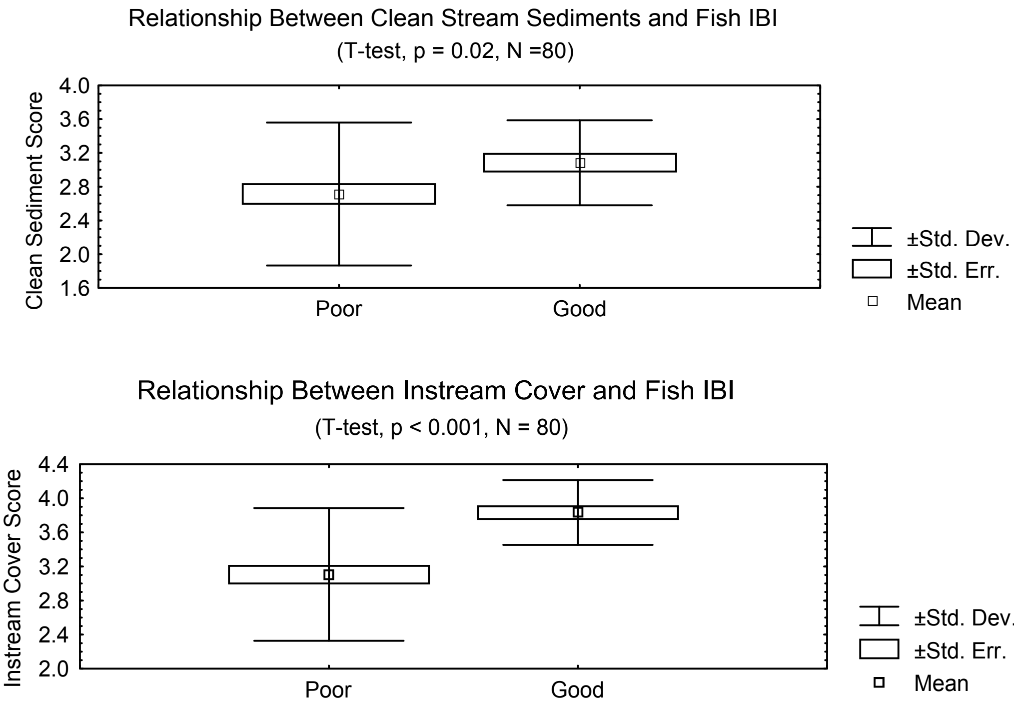


Figure 5-10. Relationship between stream embeddedness or cover and the fish IBI, categorized as either poor (impaired) or good (unimpaired), based on TVA's criteria.

5.5. CUMULATIVE EFFECTS OF LAND USE ON ASSESSMENT ENDPOINTS

Another analysis was structured to assess the cumulative effects of land use on assessment endpoints. The EPT and IBI scores were examined by subwatershed, recognizing that the type and intensity of stressors varied among subwatersheds (Table 3-3). Both the EPT and the IBI were lowest in the Guest River subwatershed (ANOVA, $p<0.05$) (Figure 5-12). The EPT was also significantly lower in the upper Powell than in either the Copper Creek or the upper Clinch River subwatersheds (ANOVA, $p<0.05$) (Figure 5-12). The IBI was not significantly different for these three subwatersheds (Figure 5-12). The Guest River has had intense coal mining activity and acid mine drainage for many years and few other land-use stressors, such as urban or pasture influences. The upper Powell subwatershed is similar, although there is slightly more urban area than in the Guest River subwatershed. Thus, coal mining activities appear to have had the greatest influence on insect and fish abundance and distribution in the Clinch and Powell watershed as a whole.

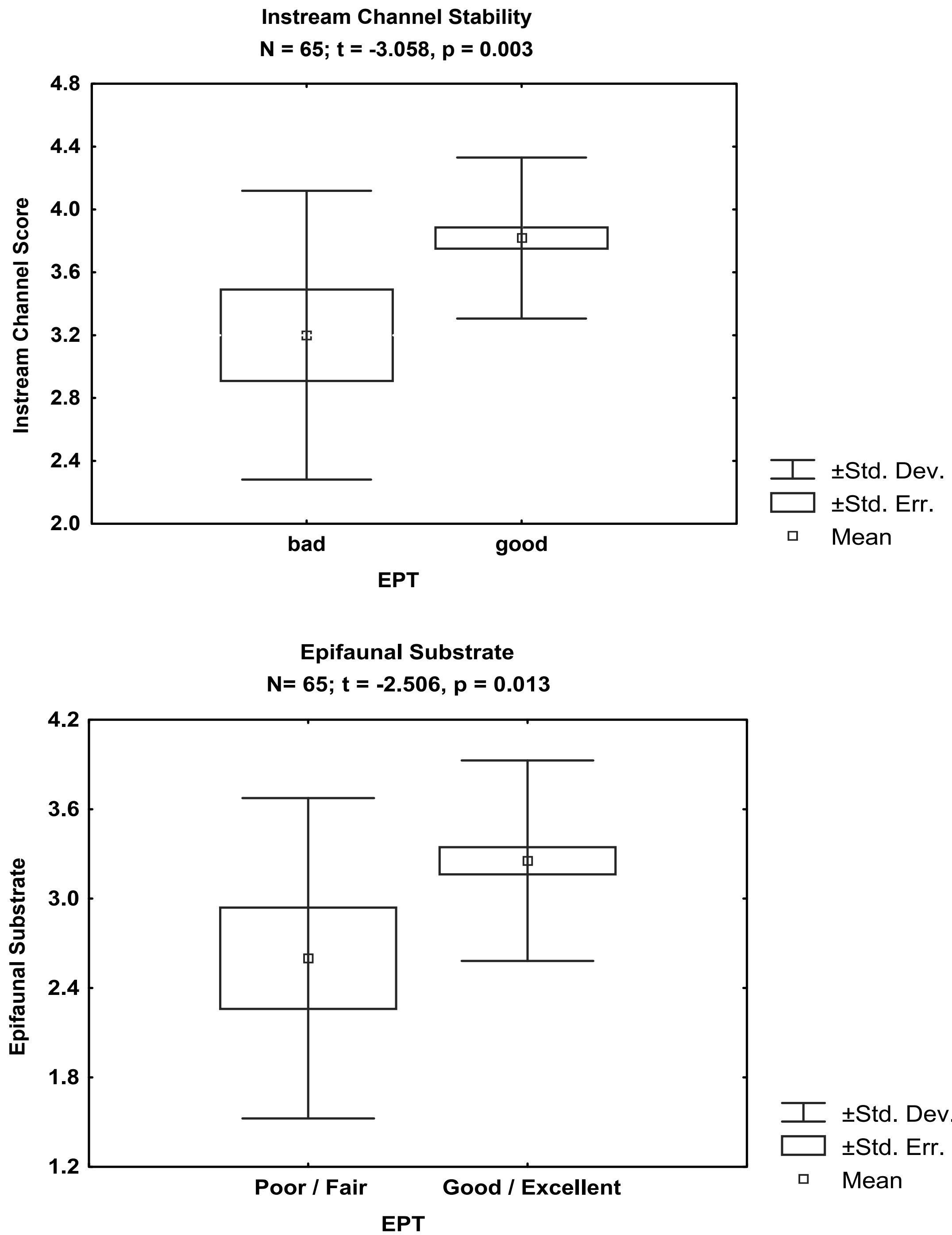


Figure 5-11. Significant relationships observed between specific instream habitat quality measures and the macroinvertebrate EPT index.

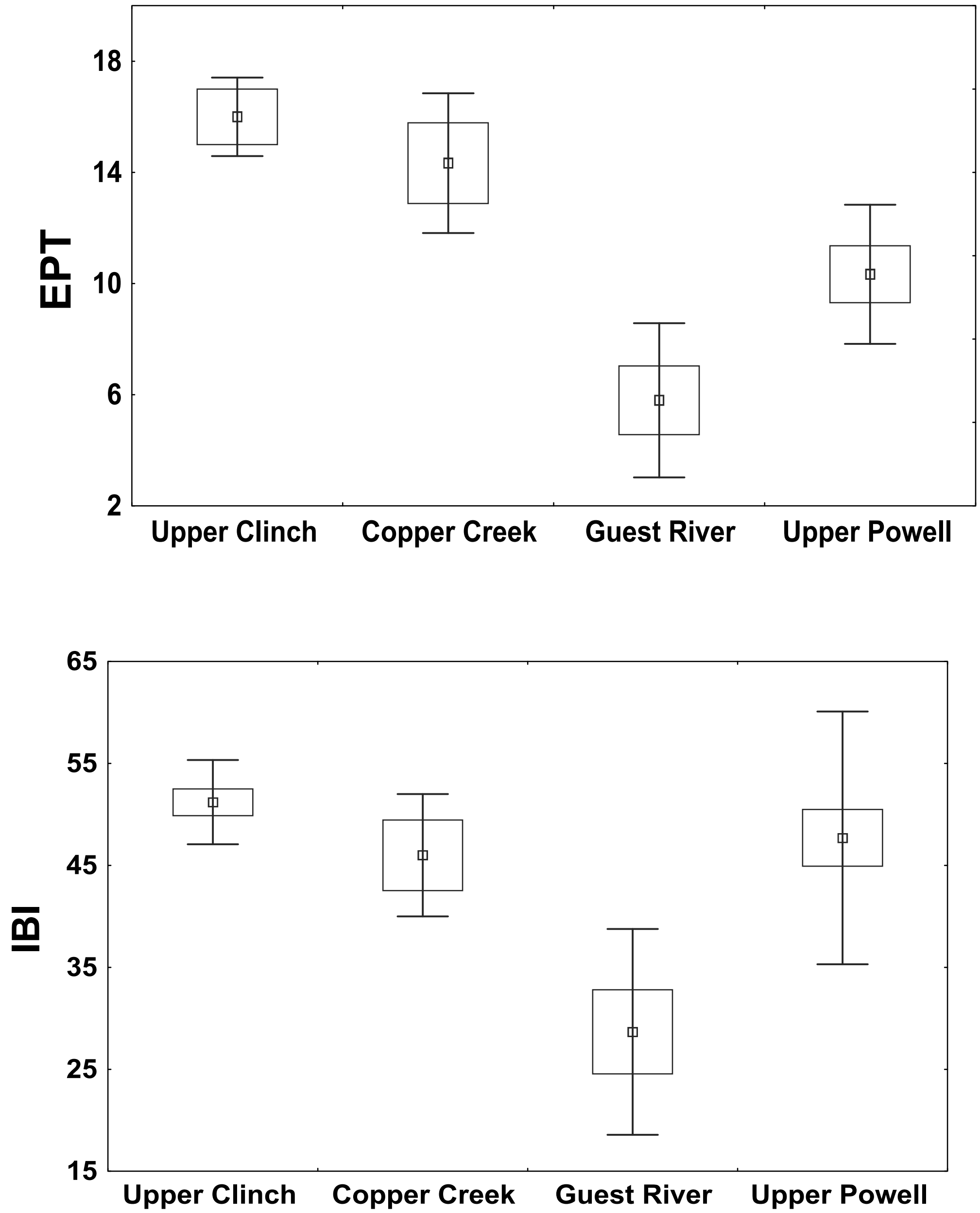


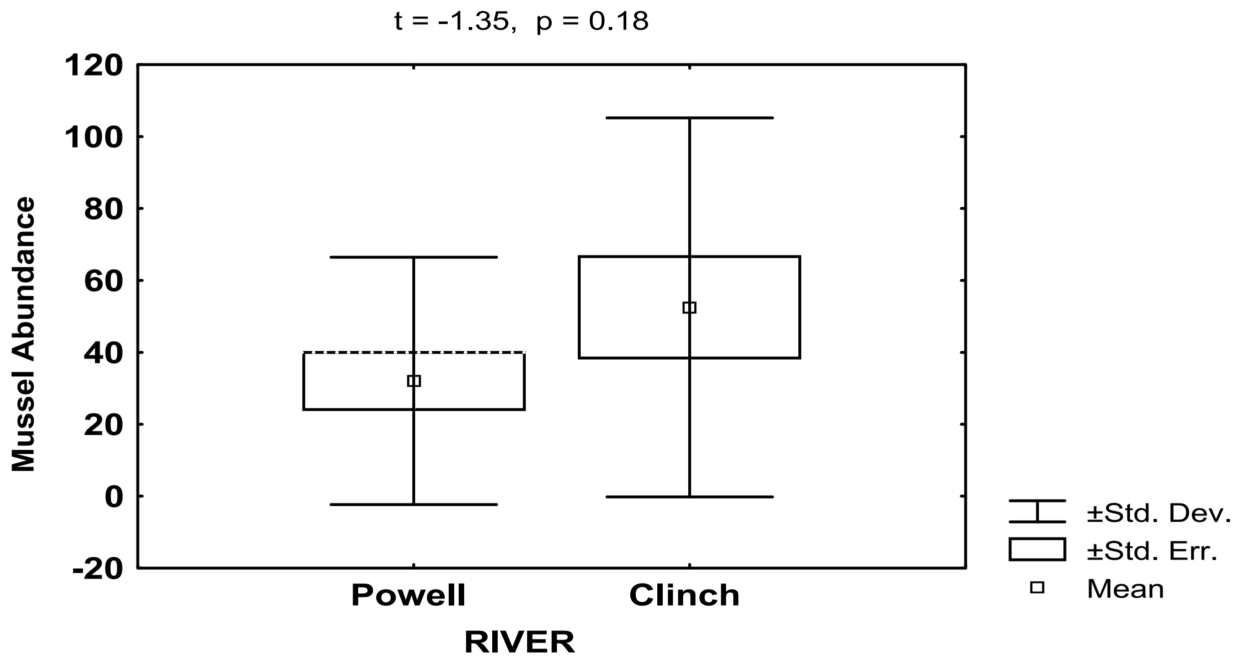
Figure 5-12. Macroinvertebrate (EPT) and fish (IBI) community integrity scores by subwatershed in the Clinch/Powell watershed based on TVA's CPRATS dataset. Boxes represent standard errors of the mean and whisker bars represent standard deviations of the mean.

Distribution and abundance data for mussels were not as readily available on a subwatershed basis as they were for insects and fish. Using the limited CMCP data for the Clinch and Powell rivers (33 sites), we observed no statistical difference in either abundance or number of species of Cumberlandian mussels between the two rivers (Figure 5-13), although species richness was close to being significantly higher in the Clinch River (p=0.07) (Figure 5-13). It is well known that most of the historic (pre-1910) mussel bed locations in both rivers have declined dramatically or been eliminated (as denoted by the reduction in red areas in Figure 5-14). Coal mining activity is one of the factors causing this pattern.

In a second type of analysis, land uses that have been determined to cause the most stress, based on either previous stepwise multiple regression analyses presented or professional experience of the workgroup, were expressed as a binary function: 0 if the particular type of land use was not present or nearby (within 2 km) and 1 if the land use was present. Prior analyses (Figure 5-7) demonstrated that gradients of effects based on proximity to land uses were not readily apparent but that threshold responses were. Table 5-5 summarizes the criteria used to designate a 0 or a 1 for those selected land uses causing the most adverse impacts. All criteria were based on results observed previously in risk analyses using the CPRATS data summarized in Figure 5-4 and Table 5-3.

Using the criteria in Table 5-5, we then computed a cumulative stressor index for each site. This index ranged from 0 to 4 because four significant sources of stress were considered. Because of the paucity of mussel data in the Guest River, only CMCP data from the Clinch and Powell rivers were used in this analysis. t-Test analysis indicated that the cumulative stressor index was greater in the Clinch River than in the Powell River (Figure 5-15) ($t = -2.24$, $p<0.05$) because of the greater frequency of urban/industrial sources and U.S. highways near the Clinch River. Mining is the chief source of stress in the upper Powell. In the Clinch River, the cumulative stress index increased as one progressed upstream ($N = 14$, $r = 0.74$, $p<0.05$)(Figure 5-16) because of more mining and urban influences in the upstream part of the river. A direct relationship between number of stressors and river mile was not observed in the Powell River ($N = 19$, $r = 0.03$, $p>0.10$), probably because of the concentration of mining activity in that subwatershed.

We observed an inverse relationship between the cumulative number of stressors and either the fish IBI or the maximum number of mussel species present at a site (Figure 5-17). Sites having two or more of the four stressors listed in Table 5-5 had greater than a 90% probability of having impaired fish community integrity and fewer than two mussel species present. Sites with one or no sources of stress had between 4 and 10 mussel species on average and, generally, an unimpaired fish community (Figure 5-17). The maximum number of mussel species observed in



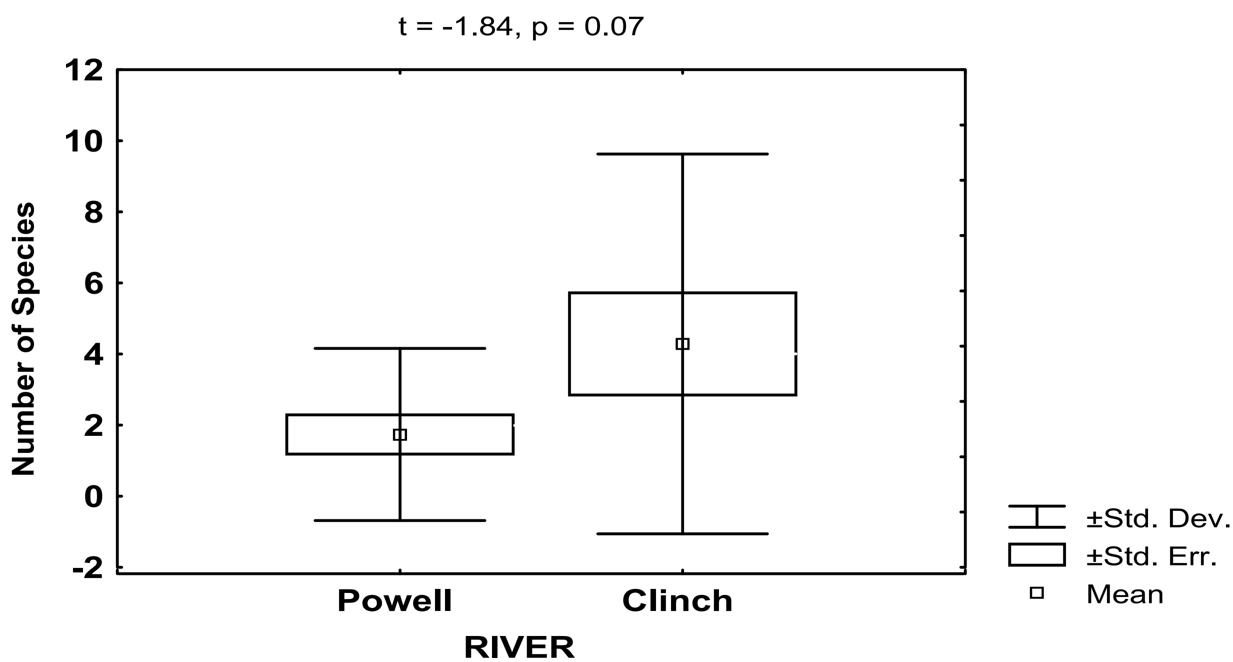


Figure 5-13. Comparison of mean number of native mussels or number of species observed in the upper Powell and Clinch rivers. Data are from TVA's CMCP survey (TVA, 1981).

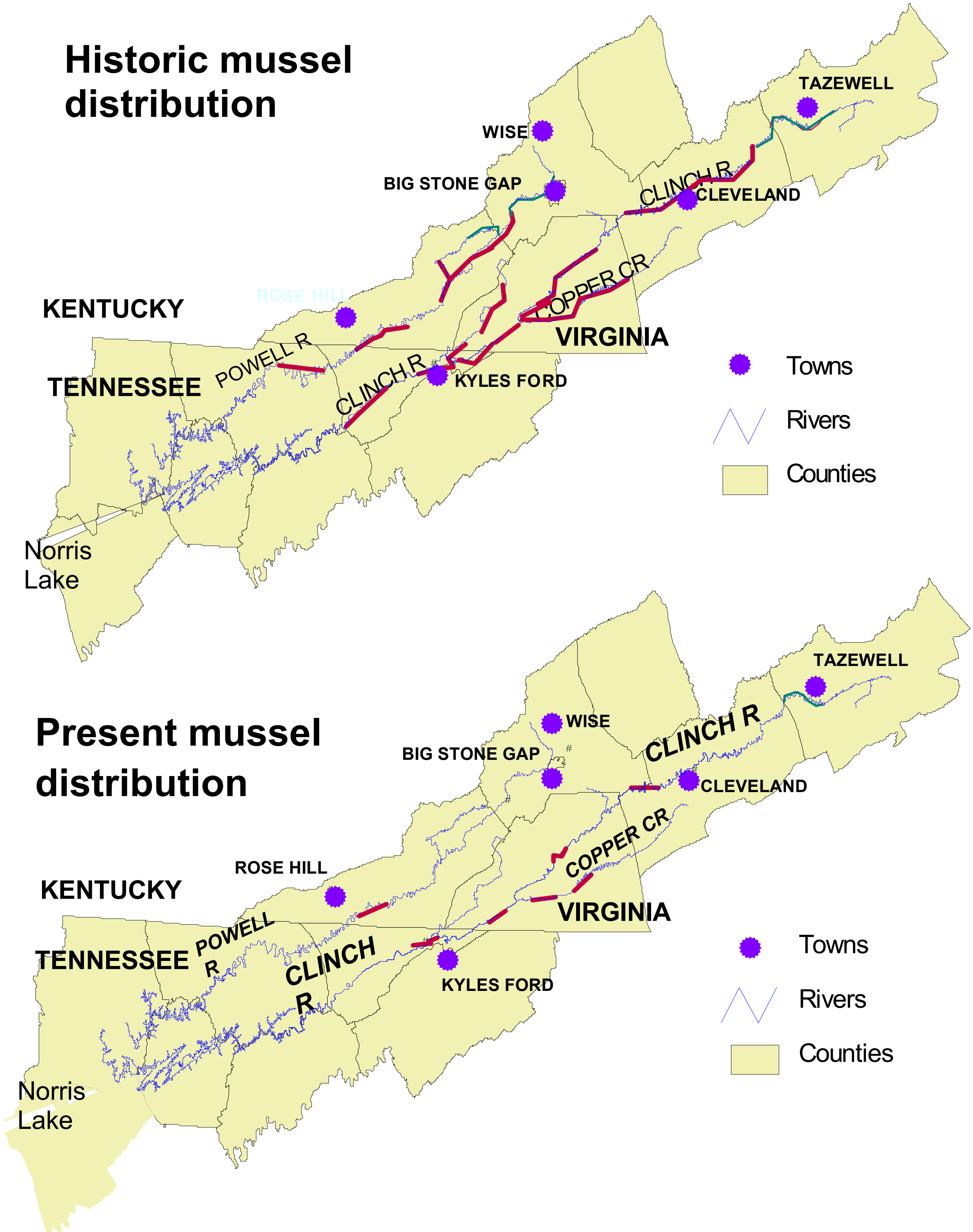


Figure 5-14. Comparison between historic (pre-1910) and present locations of native mussel concentrations in the Clinch and Powell watershed. Red areas indicate mussel beds.

Table 5-5. Criteria used to define whether a stressor was present or potentially present at a site (code = 1) or not present (code = 0) for the mussel CMCP risk analyses

Land use	Criteria
Mining	site >2 km from active mining or coal processing upstream = 0; otherwise = 1
Cropland	site >10% cropland = 1; otherwise = 0
Urban	site ≥10% urban = 1; otherwise = 0
Roads	site >2 km from road = 0; otherwise = 1

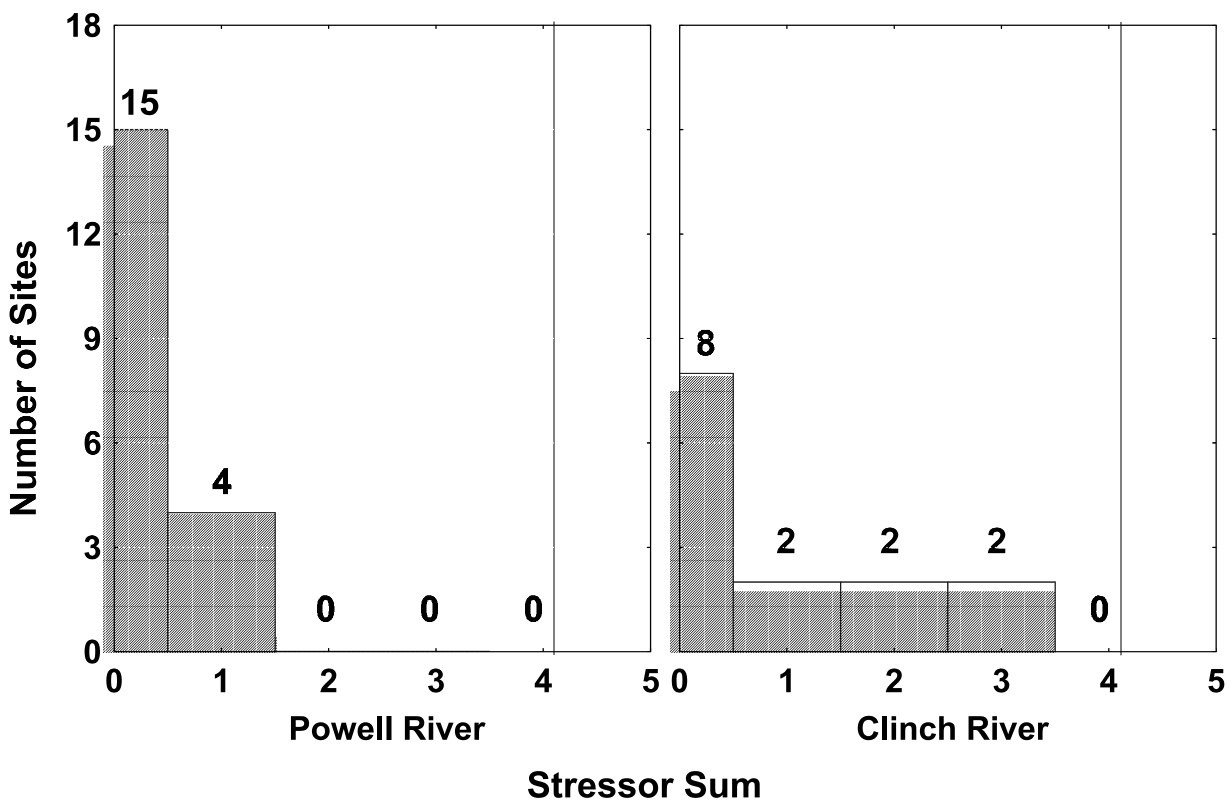


Figure 5-15. Comparison of cumulative stressors in the upper Clinch and Powell rivers. A higher index indicates more stressors present.

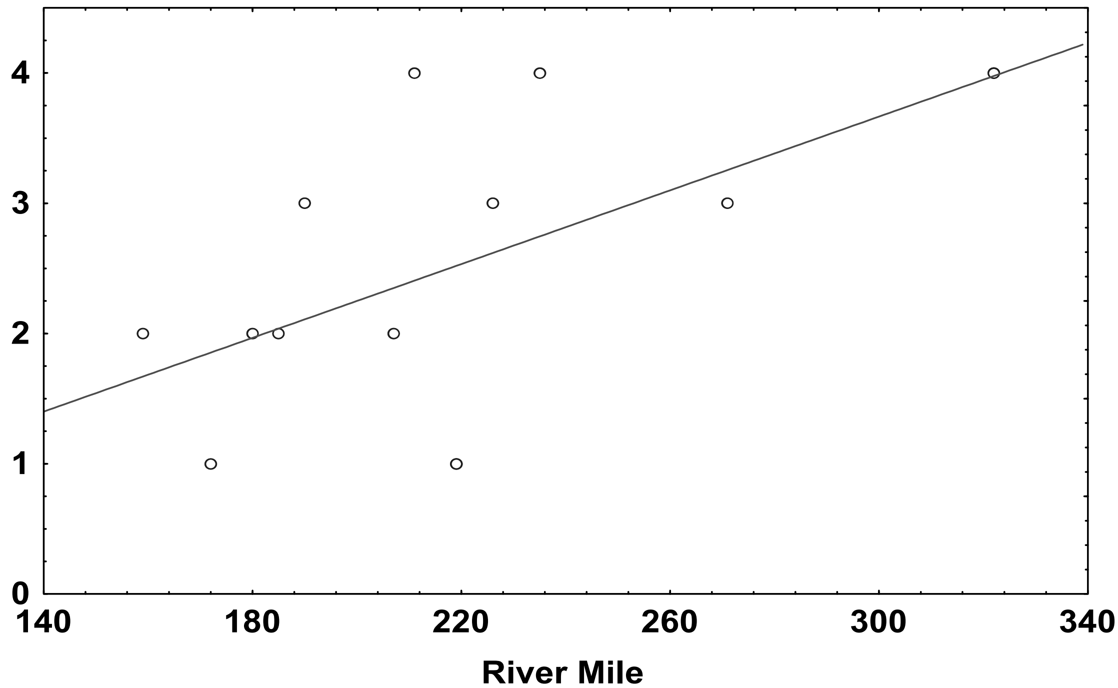


Figure 5-16. Mean number of cumulative sources of stress as one progresses upstream in the Clinch River (r = 0.74).

this dataset was 18, which is comparable to some of the better mussel sites in the entire Clinch and Powell watershed, though still far less than the historical number of species reported (> 35 species at many sites; Ortmann, 1918). This result, combined with the previous observation that fish and mussel species richness decreased with increased elevation (i.e., further upstream) even in the absence of significant sources of stress (because of the species-drainage area relationships discussed previously), suggests that mussels, fish, and perhaps other aquatic life are especially vulnerable in upstream reaches of the Powell and Clinch rivers. This pattern has also been evidenced by the fact that many current threatened and endangered mussel species were historically present in fair numbers in small tributaries and headwater areas of the Clinch and Powell watershed (Figure 5-14) (Ortmann, 1918; Ahlstedt, 1991). As discussed previously, episodic chemical spills, physical habitat degradation, and riparian corridor impairment would be expected to have greater effects in headwater areas or small tributaries in the watershed, where dilution and stream size are much reduced compared to mainstem areas.

In a preliminary attempt to examine potential exposure and vulnerability of native mussels to multiple stressors identified in this risk assessment, we used the GIS in ArcInfo to map the sites largest mussel populations in relation to mines, major roadways, urban centers, and riparian agricultural areas. There are currently no known mussel concentration sites in the upper Powell and Guest Rivers, consistent with the more intensive coal mining in that area. Figure 5-18 shows

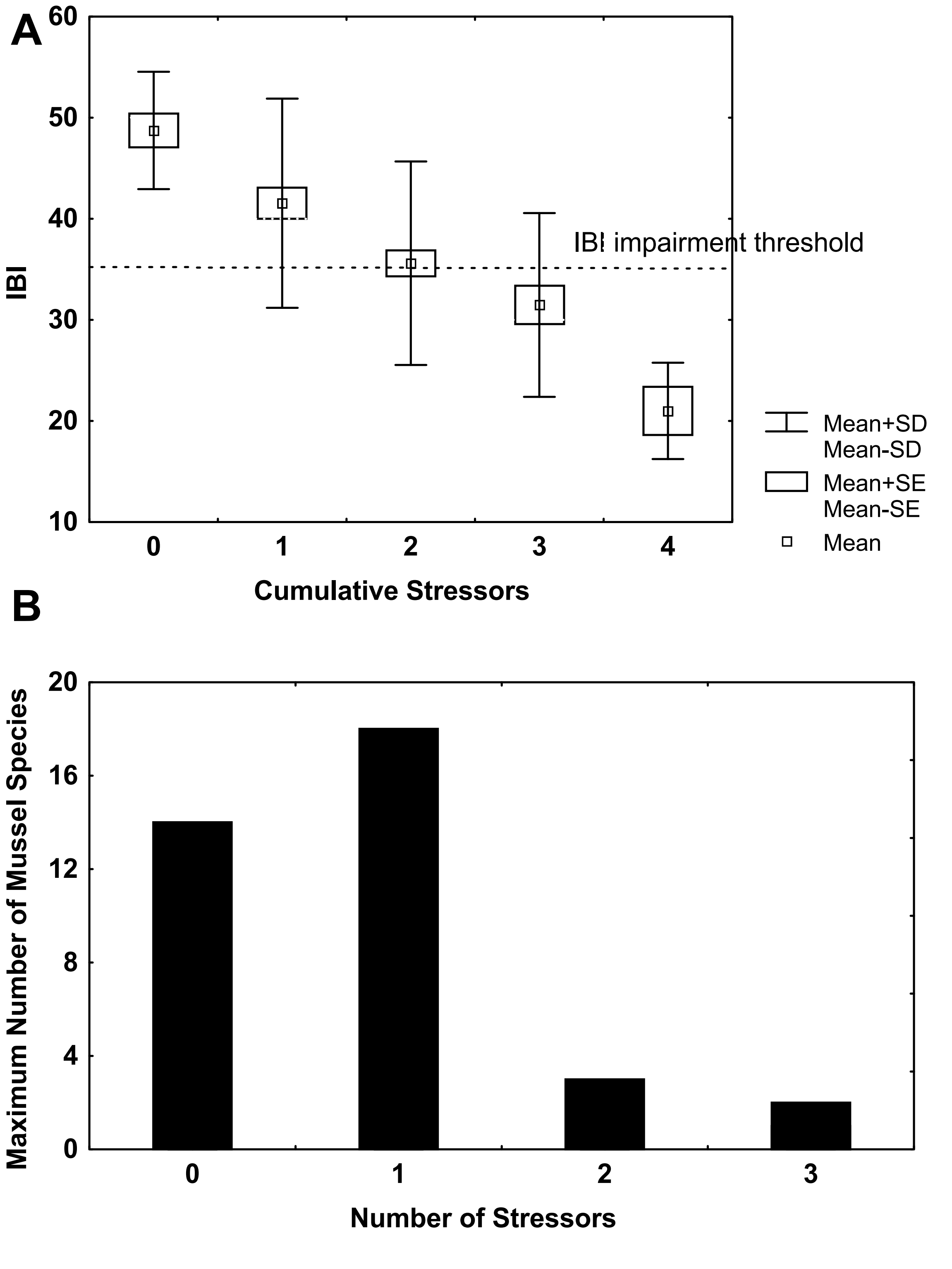


Figure 5-17. Fish IBI (a) and maximum number of mussel species collected (B) in the Clinch and Powell watershed as a function of the number of stressors present.

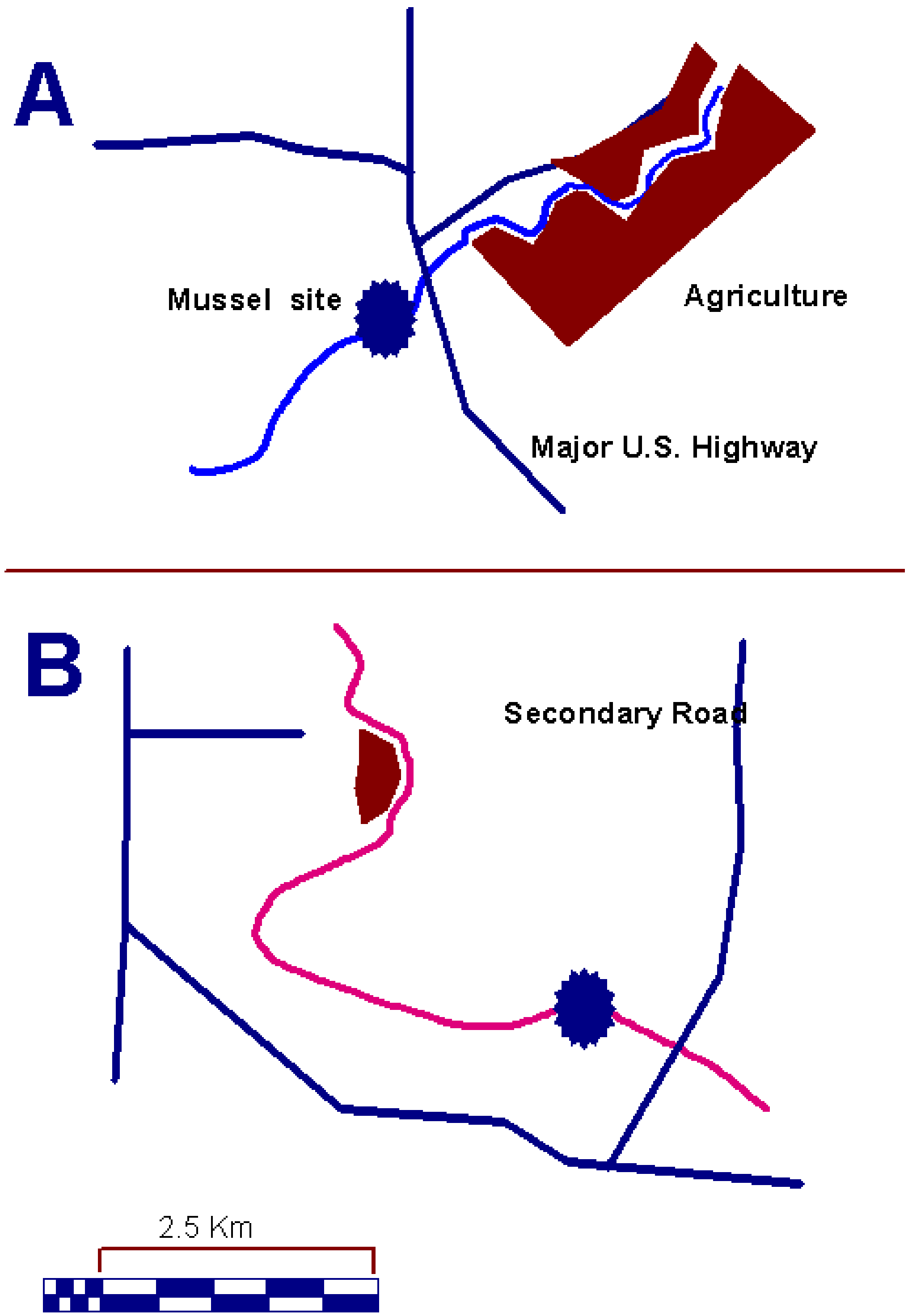


Figure 5-18. Map of portion of the middle Clinch River watershed showing two mussel concentration sites, Pendleton Island (A) and Burtons Ford (B), in relation to major roadways (a source of episodic spills) and agricultural areas.

how two examples of known mussel concentrations in the watershed with associated land-use factors analyzed this risk assessment. For example, Pendleton Island is adjacent to a major U.S. highway and directly downstream of significant agricultural riparian areas. However, other mussel sites, such as Burtons Ford, appear to be located farther away from these stressors and therefore may be less vulnerable to risk. Of the 10 mussel concentration sites examined, only about half appear to be reasonably isolated from major roads, mines, and agricultural areas. This information suggests that native mussel populations are relatively vulnerable to risks in this watershed and that further extinctions or extirpations are likely to occur unless considerable resource protection measures are taken.

5.6. SENSITIVITY ANALYSES OF RIPARIAN CORRIDOR BUFFERS

The fixed riparian corridor dimensions used in the previous risk analyses (2 km length upstream of each sampling point and 100 m to either side of the stream regardless of stream size) were recognized as a major potential source of uncertainty. Riparian zone condition may have different effects on stream biota, depending on any number of natural variations in landscape and stream characteristics. In this section we examine the sensitivity of riparian buffer dimension effects by assessing relationships between land use and biotic communities (mussels and fish) using a variety of riparian and upland buffer dimensions in the context of natural landscape and hydrologic variations. Additionally, we assess the importance of natural morphological and geographical variations to the condition of mussel and fish assemblages throughout the watershed and how these variations are related to anthropogenic influences.

5.6.1. Methods

5.6.1.1. *Mussels*

We assessed relationships among mussel abundance and richness, elevation, slope, and morphological data from 49 sites in the upper Clinch River watershed (Tazewell County) using data collected by Jones et al. (2000). From this initial list of sites, a subset (15) was randomly selected for analysis of upstream land use. Numerous sites were close to one another, especially on the mainstem, so a random selection of sites was used to eliminate redundancy in upstream percentages within these drainage areas and within riparian corridors of variable widths and lengths (100 and 200 m wide; 1, 2, 5, and 10 km long). Pearson product moment correlations ($p<0.10$) were used to test relationships between buffer characterization and mussel species richness or abundance. Qualitative analysis of mine and road locations as well as field observations reported by Jones et al. (2000) concerning site-specific stressors (e.g., open sewage pipes, oil, etc.) were also used to explain variations in broadscale landscape relationships with

mussel data. Mussel collection methods may be a source of data uncertainty because sample area and effort were not standardized across sites.

5.6.1.2. *Fish*

From the initial suite of 155 CPRATS sites from the entire watershed, 30 outside the Copper Creek subwatershed were randomly selected for analysis of riparian land-use relationships with fish IBI scores, elevation, slope, drainage area, and habitat data. Small headwater and large mainstem sites were removed from the random selection process in an attempt to reduce natural variability among sites. GIS methods and corridor sizes were the same as those used for the mussel data analysis. Pearson product moment correlations ($p<0.10$) were used to test relationships among buffer characterization, fish IBI scores, overall habitat quality scores, and 10 individual habitat parameters (see Table 3-2). IBI data were available for 29 of the 30 sites and habitat data were available for 16 sites.

5.6.2. Results and Discussion

5.6.2.1. *Mussels*

Initial analyses of land-use percentages within whole drainage areas of the 15 randomly selected sites indicated that there was little correlation between any land-use type and mussel abundance or richness (Table 5-6). Riparian land use, however, was correlated with mussel richness (Table 5-6). The proportion of forested land in all corridor dimensions was positively and significantly correlated with mussel species richness, with the highest correlation occurring within 100-m-wide and 5-km-long corridors (Table 5-6 and Figure 5-19). The amount of urban land within riparian corridors appears to adversely affect mussel richness, with 2- and 5-km-long corridors showing the highest correlation with the number of mussel species at a site. Pasture land in 100-m-wide, 5-km-long corridors was inversely correlated with mussel richness.

Streams were grouped into different classes of order, slope, and elevation to assess the importance of riparian zone dimensions to mussel assemblages. Analyses of riparian corridor dimensions in high-order (> 4^{th} order) and low-order (3^{rd} or 4^{th} order) streams yielded little distinction between the two site classes in terms of relationships with mussel fauna characteristics (i.e., 100 m × 5 km corridors for high- and low-order streams had the highest correlation with mussel richness). Elevation was also investigated as a possible mediating factor, based on analyses presented in section 5.1, but it did not show any distinctive relationships with land use and mussel richness.

Table 5-6. Correlation (Pearson product moment R values) of land-use percentages within whole drainage areas and within different-size riparian corridors with mussel richness and abundance[a]

Riparian corridor	Land use	Mussel richness[b]	Mussel abundance
Whole drainage basin	Forest	0.12	-0.20
	Urban	0.35	-0.03
	Pasture	-0.20	0.20
100 m x 1 km	Forest	**0.62**	0.11
	Urban	-0.46	-0.05
	Pasture	-0.33	-0.07
100 m x 2 km	Forest	**0.68**	-0.19
	Urban	**-0.49**	-0.06
	Pasture	-0.39	0.25
100 m x 5 km	Forest	**0.77**	-0.07
	Urban	-0.40	-0.13
	Pasture	**-0.59**	0.18
100 m x 10 km	Forest	**0.65**	0.07
	Urban	-0.08	-0.14
	Pasture	0.15	-0.22
200 m x 1 km	Forest	**0.62**	0.11
	Urban	-0.43	-0.04
	Pasture	-0.39	-0.08
200 m x 2 km	Forest	**0.69**	-0.16
	Urban	-0.46	-0.01
	Pasture	-0.43	0.21
200 m x 5 km	Forest	**0.62**	-0.16
	Urban	-0.28	**-0.05**
	Pasture	-0.47	0.19
200 m x 10 km	Forest	**0.62**	0.05
	Urban	0.01	-0.08
	Pasture	0.19	-0.21

[a]Bold values are significant at $p<0.10$

[b]N = 13; two outliers removed because of the site-specific factors

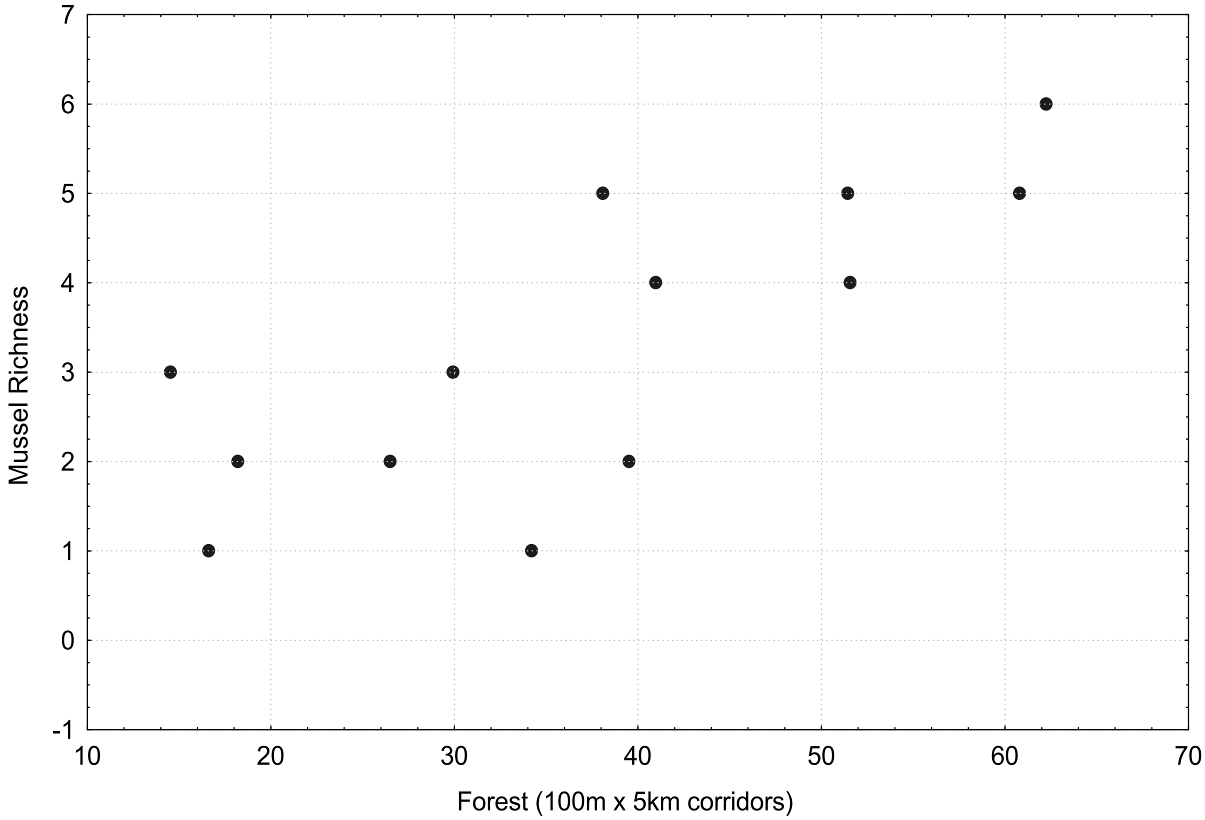


Figure 5-19. Mussel richness versus percent forested land use in 100-m-wide, 5-km-long riparian corridors.

Average catchment slope was the only landscape factor that appeared to have an affect on which corridor dimensions were most correlated with mussel richness. Using a slope of 33% as a threshold (because it provided a relatively even split of the 13 sample sites used for this analysis, two sites having been removed because of site-specific factors), the amount of forested land within 100 m × 5 km corridors appeared to be the best predictor of mussel richness for streams with > 33% average slope (Table 5-6; Appendix C, Figure C-1). However, in streams with average catchment slopes < 33%, shorter corridors (1 km) seemed to show a stronger relationship with mussel richness (Appendix C, Figures C-1 to C-3).

The above results suggest that streams with higher slopes may require longer upstream forested riparian areas than low-slope streams because of the greater influence that runoff may have in higher-gradient areas. These results support those described for Copper Creek risk analyses (Chapter 4). Land use in 1-km-long corridors upstream of the sampling point was the best predictor of mussel characteristics in both the low-catchment-slope Tazewell County sites and in the Copper Creek sites, most of which had slopes similar to those of the low-slope Tazewell County sites. These results also suggest that using a 2-km-length corridor upstream of

each sampling point throughout the watershed may have underestimated land-use effects in higher gradient sites. Although these data yielded indications of riparian corridor sizes that may be most efficient for predicting mussel fauna characteristics, the results should be used cautiously because of the limited number of data points.

Although land-use relationships with mussel species richness were observed, these patterns may also be due to natural variability in stream characteristics, specifically stream order and elevation. Low-elevation or high-order stream sites had more mussel species than high-elevation or low-order sites (t-test, $p<0.05$) (Appendix C, Figures C-3 and C-4). However, as both elevation and stream order were correlated with proportion of riparian forest ($r = -0.76$, $p<0.05$ [elevation vs. forest within a 100 m × 5 km riparian buffer]) it is difficult to determine whether mussel species richness is influenced more by land use or by stream order/elevation. Within a given elevation or stream order group, there was little correlation between riparian forestland and mussel species richness, which suggests that mussel richness is, in fact, related more to stream size or elevation than to riparian land use. Furthermore, the small number of sites that constitute test groups in our analyses makes it difficult to statistically assess these alternative hypotheses. Finally, the variability in land-use percentages may not be wide enough in this dataset to allow broad-scale relationships to be distinguished.

Correlation analysis indicated no significant relationships between land use and mussel abundance (Table 5-6). This observation may be due to natural differences between smaller and larger streams in regard to mussel abundance. Small (3rd or 4th order) and large (> 4th order) streams were analyzed separately, and relationships between land use and mussel abundance were, in fact, observed. High-order streams that had more than 50% forested land within the riparian buffer—defined as 100 m wide and 1–2 km long—had significantly more mussels (median = 68) than sites with less than 50% riparian forest, (median = 23; t-test, $p<0.05$) (Appendix C, Figure
C-5). Relationships between land use and mussel abundance were also observed in low-order streams; however, the pattern was the opposite of what was expected. Streams that had more than 30% forested cover in riparian buffers 100–200 m wide and 1–2 km long had fewer mussels (median = 7) than those that had less than 30% forested riparian land (median = 197; t-test, $p<0.05$) (Appendix C, Figure C-6). Land use within longer corridors (5 and 10 km) of low-order streams showed less correlation with mussel abundance.

The inverse relationship between forested land cover and mussel abundance in low-order streams may be due to other landscape factors such as slope and elevation, which are also correlated with land use, as explained previously. Both mussel richness and abundance decreased as elevation increased (i.e., richness and abundance are lower in headwater areas than in the mainstem) (Appendix C, Figure C-7), as noted in previous analyses of both the CMCP and

CPRATS datasets (section 5.1). Average slope of the drainage area upstream of a site showed little correlation with mussel richness or abundance when all sites within the Clinch River watershed were evaluated. However, in high-order streams (i.e., sites primarily on the mainstem), mussel abundance and richness increased with slope of the drainage area (Appendix C, Figure C-8). These higher slope areas are located toward the lower end of the mainstem (Appendix C, Figure C-9), which also is more highly forested (Appendix C, Figure C-10) than the rest of the watershed. Higher-sloped sections of the mainstem may contain cleaner, less embedded substrate that is more suitable for mussel colonization than lower-slope areas, where fine sediment can more easily settle on the stream bottom, smothering gravel and cobble substrate.

In low-order (smaller) streams, the relationship between slope and mussel richness or abundance, although less pronounced, is the opposite of that observed for the mainstem (Appendix C, Figure C-11). Steep slopes in these headwater areas may result in scouring during high flows, reducing the amount of suitable mussel habitat. Furthermore, these headwater areas often contain large amounts of boulder and bedrock substrate which, although natural for these areas, may be unsuitable for mussel colonization and/or fish host distribution. Thus, knowing the drainage slope as well as elevation and stream size reduced uncertainties in predicting effects of land use on native mussel communities.

In addition to the effects of land use, elevation, and slope on mussel fauna characteristics, site-specific factors also may have significant effects. According to Jones et al. (2000), several sites appeared to be contaminated by sewage from leaks or open pipes. For example, the water at site 41 smelled of oil and, concurrently, site 49 had no mussels and all crayfish were dead, suggesting toxic inputs. These site-specific factors can be observed as outliers of some of the broad-scale patterns observed for the watershed (Appendix C, Figures C-12 and C-13). Drought conditions in some headwater streams may also limit mussel reproduction (because of the reliance on fish populations) and colonization.

5.6.2.2. *Fish*

5.6.2.2.1. ***Habitat***. Our initial results (section 5.2) indicated that there was little correlation between land use and habitat. However, when the influence of riparian land use was considered, relationships were apparent (Table 5-7). Overall habitat quality increased with the percent of riparian forest (100 and 200 m × 1 km corridor dimensions) and decreased as urban land increased (all corridor dimensions). Individual habitat parameters, including bank vegetative protection, bank stability, and riparian vegetative protection were positively correlated with forested land in 1- and 2-km-long corridors. The amount of urban land within most corridor sizes was inversely correlated with all of the individual habitat parameters except channel flow and riparian vegetation .

Table 5-7. Correlation (Pearson product moment R values) of total habitat scores and 10 individual habitat parameters with land uses in whole drainage areas and different-size riparian corridors[a]

Riparian corridor	Land use	Total habitat score	Instream cover	Epifaunal substrate	Embeddedness	Channel stability	Sedimentation	X6FRRI	Channel flow status	Bank vegetation	Bank stability	Riparian vegetation
Whole drainage basin	Forest	-0.06	0.13	-0.27	-0.10	-0.20	-0.22	-0.24	0.16	-0.08	0.01	0.31
	Urban	-0.43	-0.40	**-0.65**	-0.40	**-0.68**	-0.23	**-0.52**	0.45	-0.22	-0.19	-0.04
	Pasture	0.08	-0.12	0.30	0.10	0.23	0.22	0.26	-0.15	0.08	0.00	-0.30
100 m x 1 km	Forest	**0.62**	0.47	0.03	0.50	0.28	0.33	0.34	0.32	**0.66**	**0.72**	**0.62**
	Urban	**-0.80**	**-0.70**	**-0.81**	**-0.71**	**-0.91**	**-0.52**	**-0.84**	0.16	**-0.58**	**-0.57**	-0.21
	Pasture	-0.22	-0.13	0.37	-0.16	0.19	-0.12	0.10	-0.42	-0.39	-0.44	-0.43
100 m x 2 km	Forest	0.47	0.43	-0.05	0.44	0.13	0.29	0.15	0.31	**0.51**	**0.57**	0.40
	Urban	**-0.81**	**-0.72**	**-0.82**	**-0.73**	**-0.93**	**-0.53**	**-0.85**	0.18	**-0.58**	**-0.57**	-0.23
	Pasture	-0.12	-0.08	0.37	-0.14	0.27	-0.13	0.21	-0.41	-0.28	-0.32	-0.24
100 m x 5 km	Forest	0.19	0.21	-0.19	0.13	-0.09	0.10	-0.13	0.35	0.23	0.29	0.35
	Urban	**-0.72**	**-0.58**	**-0.83**	**-0.55**	**-0.87**	-0.41	-0.79	0.25	**-0.50**	-0.48	-0.24
	Pasture	-0.04	-0.07	0.35	-0.04	0.27	-0.05	0.28	-0.37	-0.13	-0.18	-0.25
100 m x 10 km	Forest	-0.15	-0.20	-0.12	-0.15	-0.24	-0.09	-0.32	0.40	-0.07	-0.05	-0.03
	Urban	**-0.60**	-0.43	-0.79	-0.46	**-0.75**	-0.40	**-0.65**	0.30	-0.39	-0.37	-0.19
	Pasture	-0.11	-0.13	0.29	-0.17	0.21	-0.14	0.19	-0.17	-0.17	-0.23	-0.23
200 m x 1 km	Forest	**0.57**	0.41	0.02	**0.52**	0.23	0.40	0.30	0.24	**0.59**	**0.65**	**0.54**
	Urban	**-0.82**	**-0.71**	**-0.82**	**-0.72**	**-0.92**	**-0.53**	**-0.85**	0.16	**-0.59**	**-0.58**	-0.23
	Pasture	-0.17	-0.05	0.36	-0.18	0.24	-0.19	0.14	-0.33	-0.31	-0.36	-0.32
200 m x 2 km	Forest	0.44	0.38	-0.03	0.41	0.11	0.32	0.14	0.32	0.45	**0.50**	0.38
	Urban	**-0.81**	**-0.72**	**-0.82**	**-0.73**	**-0.93**	**-0.53**	**-0.85**	0.18	**-0.57**	**-0.56**	-0.22
	Pasture	-0.10	-0.04	0.34	-0.13	0.27	-0.15	0.21	-0.39	-0.22	-0.27	-0.23
200 m x 5 km	Forest	0.16	0.16	-0.17	0.09	-0.10	0.08	-0.14	0.35	0.18	0.24	0.33
	Urban	**-0.73**	**-0.59**	**-0.85**	**-0.57**	**-0.90**	-0.43	**-0.79**	0.27	-0.49	-0.48	-0.25
	Pasture	-0.03	-0.03	0.32	-0.02	0.27	-0.04	0.28	-0.37	-0.10	-0.15	-0.24
200 m x 10 km	Forest	-0.15	-0.21	-0.10	-0.15	-0.23	-0.09	-0.31	0.37	-0.09	-0.06	-0.03
	Urban	**-0.62**	-0.46	**-0.81**	-0.49	**-0.79**	-0.42	**-0.68**	0.31	-0.39	-0.37	-0.20
	Pasture	-0.11	-0.12	0.27	-0.17	0.20	-0.14	0.17	-0.15	-0.17	-0.22	-0.24

[a]Bold values are significant at $p<0.10$ (N = 12)

To assess whether natural landscape variability influenced the way in which riparian land use affected habitat, stream habitat data were analyzed according to drainage area, elevation, and slope categories. For smaller streams (< 50 km^2 drainage) and high-elevation streams (> 450 m) (N = 7), additional land use/habitat relationships were apparent ($p<0.10$): pasture within most riparian corridors was inversely correlated with overall habitat scores (r~–0.9) as well as channel flow (r~–0.8), bank vegetation (r~–0.7) and bank stability (r~–0.8), and riparian protection (r~–0.8). Pastureland in 2-km-long corridors showed an inverse correlation with sedimentation

scores (i.e., sediment deposits increased with upstream pasture land) (r~–0.75). For larger streams (> 50 km^2) and lower-elevation streams (< 450 m) (N = 5), nearstream land use did not predict habitat quality. For streams with average percent catchment slopes > 36% (N = 7), development was inversely correlated with all habitat parameters (r = –0.68 to –0.91, p<0.10) except sedimentation, channel flow status, and riparian vegetation. In lower-sloped streams (< 36% slope) (N = 5), the same habitat scores were more closely correlated (inversely) with land use in 1- and 2-km-long corridors. When compared with our previous findings (section 5.2), these results suggest that riparian land use may influence stream habitat more than land use within the entire drainage area, and that habitats in smaller, higher-elevation streams may exhibit more riparian land-use effects than larger, lower-elevation streams. Because of limited data, however, it is difficult to assess the riparian corridor dimensions that exhibit the greatest influence on habitat quality. Furthermore, the results described above should be used cautiously because of the small amount of data available for analyses.

5.6.2.2.2. ***IBI.*** Various riparian land uses were correlated with IBI scores (Table 5-8). In all corridor sizes, pasture and urban land were correlated (positively and inversely, respectively) with IBI scores. Forested land in 5- and 10-km-long corridors was inversely correlated with IBI. Relationships between forested and pasturelands and IBI scores are opposite those normally expected for these types of land uses. These results are similar to those described in Section 5.3, which emphasized the influence of mining (found mostly in forested areas) on IBI scores. IBI scores decreased as average catchment slope increased (Figure 5-20). To reduce confounding factors, streams with slopes > 36 and < 36% were investigated separately. For streams with slopes > 36%, IBI scores decreased as riparian urban land increased (all corridor sizes) (Table 5-8). For streams with slopes < 36%, only urban land within longer corridors (5 and 10 km) and within whole catchments seemed to influence IBI scores (Table 5-8). The percentage of forested land and the average catchment slope are both related to the number of mines in a region (Figure 5-21). Therefore, investigating high- and low-slope areas separately allows assessment of landscape factors not correlated with slope. In fact, urban land was the only land-use correlated with IBI after slope was standardized (and presumably the influence of mining was less of a factor), indicating that urban land was the only land use with a detectable influence on IBI scores.

Elevation was investigated as a possible factor affecting the way in which riparian corridor characteristics influence IBI scores, as noted in section 5.1. For streams with minimum elevations of > 370 m, IBI scores increased with nearstream pasture and cropland and decreased as urban land increased (Table 5-8). Within entire drainages of streams in this elevation category, forested land was inversely correlated with IBI scores. Land-use correlations were less evident for lower-elevation streams (minimum elevations < 370 m). These observations seem to further

Table 5-8. Correlation (Pearson product moment R values) of IBI scores with whole drainage and riparian corridor land uses for all sites and various categories, based on slope, elevation, and drainage area.[a]

Riparian corridor	Land use	IBI score (n=22)	IBI Score (slope >36; n=12)	IBI score (slope <36; n=10)	IBI score (site elevation >370 m) (n=12)	IBI score (site elevation <370 m) (n=10)	IBI score (drainage area >50 sq km) (n=10)	IBI score (drainage area <50 sq km) (n=12)
100 m x 1 km	Forest	-0.15	-0.05	-0.13	-0.35	0.07	-0.09	-0.21
	Urban	**-0.54**	**-0.57**	-0.54	**-0.56**	-0.47	**-0.69**	-0.38
	Pasture	**0.46**	0.49	0.39	**0.65**	0.26	0.51	0.43
100 m x 2 km	Forest	-0.16	0.03	-0.11	-0.37	0.04	-0.01	-0.28
	Urban	**-0.45**	**-0.55**	-0.42	**-0.56**	-0.35	**-0.65**	-0.24
	Pasture	**0.43**	0.35	0.34	**0.61**	0.25	0.40	0.47
100 m x 5 km	Forest	**-0.39**	-0.22	-0.07	-0.42	-0.38	-0.39	-0.42
	Urban	**-0.50**	**-0.60**	**-0.63**	**-0.51**	-0.40	**-0.67**	-0.33
	Pasture	**0.49**	0.36	0.28	**0.55**	0.43	0.48	**0.51**
100 m x 10 km	Forest	**-0.42**	-0.44	-0.15	-0.46	-0.47	-0.46	**-0.42**
	Urban	**-0.38**	**-0.54**	-0.49	-0.42	-0.22	**-0.68**	-0.23
	Pasture	**0.42**	0.24	0.24	**0.52**	0.36	0.51	0.41
200 m x 1 km	Forest	-0.21	-0.09	-0.23	-0.43	0.04	-0.26	-0.22
	Urban	**-0.52**	**-0.54**	-0.53	**-0.55**	-0.47	**-0.64**	-0.40
	Pasture	**0.51**	**0.51**	0.45	**0.68**	0.36	**0.61**	0.45
200 m x 2 km	Forest	-0.16	0.01	-0.07	-0.37	0.10	-0.08	-0.27
	Urban	**-0.44**	-0.48	-0.47	**-0.56**	-0.40	**-0.57**	-0.26
	Pasture	**0.46**	0.39	0.32	**0.61**	0.31	0.48	0.46
200 m x 5 km	Forest	**-0.45**	-0.24	-0.10	-0.44	-0.44	-0.48	-0.44
	Urban	**-0.48**	**-0.57**	**-0.66**	**-0.50**	-0.40	**-0.63**	-0.34
	Pasture	**0.53**	0.37	0.32	**0.56**	0.49	0.54	**0.54**
200 m x 10 km	Forest	**-0.44**	-0.45	-0.17	-0.46	-0.50	-0.51	-0.42
	Urban	**-0.40**	**-0.53**	-0.56	-0.44	-0.24	**-0.64**	-0.27
	Pasture	**0.45**	0.25	0.25	**0.53**	0.39	**0.55**	0.41
Whole Drainage	Forest	**-0.55**	-0.31	-0.13	**-0.61**	-0.44	-0.54	**-0.60**
	Urban	**-0.39**	**-0.51**	**-0.62**	-0.44	-0.32	**-0.65**	-0.16
	Pasture	**0.59**	0.38	0.21	**0.66**	0.47	**0.55**	**0.67**

[a]Bold values are significant at $p<0.10$

illustrate the influence of mining on fish assemblage status. Because mining in this region is found mostly in high-elevation forested areas (Figure 5-21), effects from mining may explain why in these higher-elevation forested and pasture areas exhibit correlations with streams that are the opposite of what is normally expected.

Streams of different drainage sizes were separated to further investigate IBI relationships with riparian landscape characteristics. In large streams (catchment size > 50 km^2), IBI scores decreased as nearstream and whole-drainage urban land increased, whereas in smaller streams

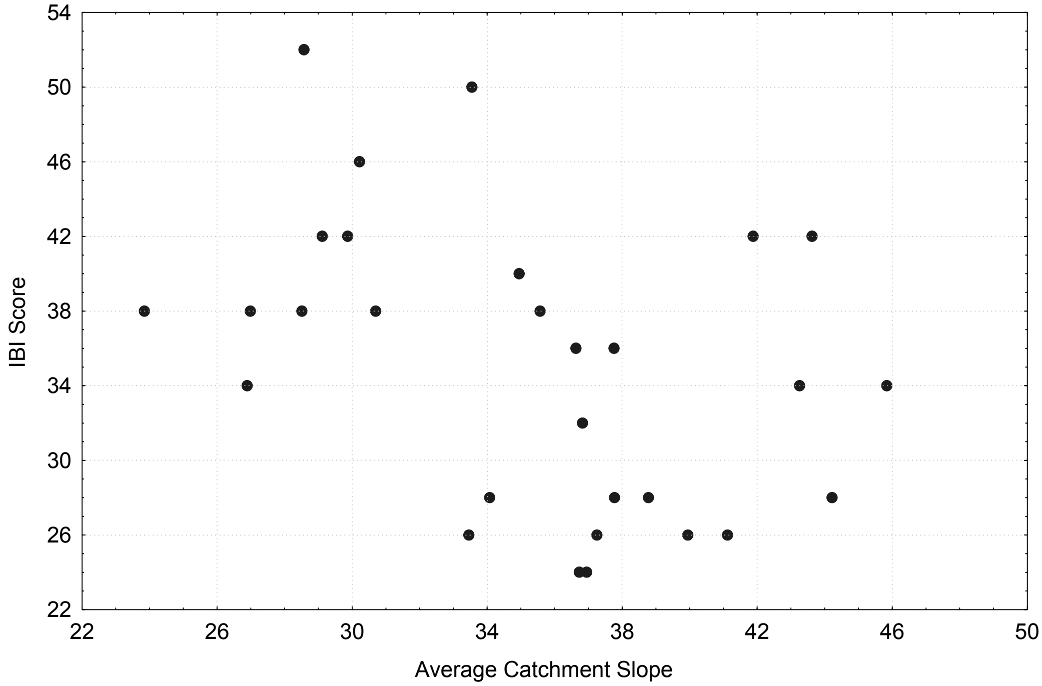


Figure 5-20. IBI score versus average catchment slope (%).

(catchment size < 50 km^2), urban land was not significantly correlated with IBI scores (Table 5-8). In these smaller streams, pasture land in 5-km corridors and whole-drainage areas was positively correlated with IBI scores. Again, this observation is most likely due to an association of particular land uses with mining activities; pastureland is inversely related to mines. Average catchment slope within these different-sized drainage areas also was predictive of IBI scores. For large streams, IBI scores were inversely correlated with average catchment slope ($r = -0.79$; $N = 14$). When elevation is considered, the correlation becomes even stronger: IBI scores of large streams with average elevations of < 550 m were closely correlated with average catchment slope ($r = -0.90$; $N = 9$). The correlation of slope with IBI scores for small streams was insignificant.

A confounding factor in the relationship of slope with larger stream IBI scores, however, is that slope is also correlated with forest, pasture, and cropland in these areas. An important point here is that previous results (section 5.3) have shown that these land uses are not the primary factors affecting IBI scores; mining in these particular land-use areas was determined to have the most influence on IBI scores. Although slope was correlated with IBI scores in large streams, this is most likely due to the greater presence of mining in steeper, more highly forested areas, as described in section 5.3, and not to any direct effects of slope on fish assemblages. Urban land, however, was not correlated with mining activities, indicating that urban land does in fact affect IBI scores in large streams. Although our data do not demonstrate an influence of urban land

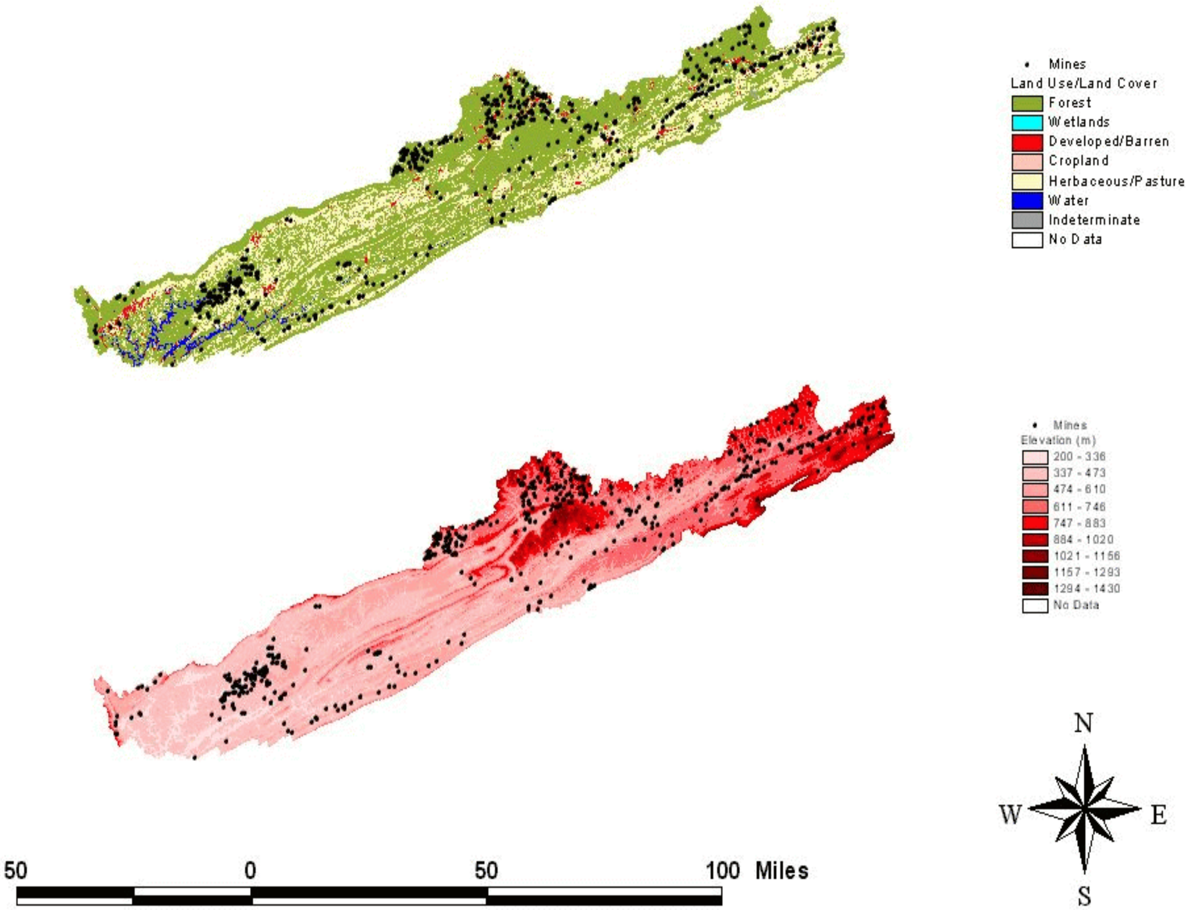


Figure 5-21. Mines in the Clinch River watershed. Green areas indicate forest in the upper figure; dark red areas in the lower figure indicate high-slope areas.

on smaller streams, any relationships may be confounded by the greater abundance of mines in upland areas, where these streams are more often located.

In summary, results of additional analyses examining the effects of riparian corridor dimension on relationships between biota and land use generally support those obtained using Copper Creek information (Chapter 4). However, for smaller, high-gradient (i.e., generally high-elevation) stream locations (which represent < 5% of the total sites examined in the CPRATS dataset), a shorter riparian corridor length (1 km rather than 2 km, as used in our risk analyses) would have yielded more accurate results concerning land-use effects on biota at those sites. Thus, for many of the sampling locations examined in risk analyses in Chapter 5 (including TVA's CPRATS dataset), there appears to be a moderate degree of certainty in our results.

6. RISK CHARACTERIZATION

6.1. FINDINGS

6.1.1. Copper Creek Watershed

The pilot study conducted in Copper Creek to refine the methodological approach for the entire watershed assessment was also useful in describing the cause of problems in that subwatershed. It is likely that embeddedness and sedimentation affect the abundance and distribution of invertebrate and fish species in Copper Creek, as both habitat characteristics were directly related to the amount of agricultural land use in the riparian corridor and the measures of biota (Figures 4-4 and 4-5). Optimal benefits to fish, mussels, and perhaps other invertebrates would be realized by maintaining the riparian corridor for 500 to 1500 m upstream and for 100 m on either side of the stream for the site of interest (Figure 4-3). Fish taxa richness was found to be a useful surrogate measure for mussel species richness where mussel data were lacking (Figure 4-11).

6.1.2. Clinch and Powell Watershed

Relationships between land use and habitat quality in the Clinch and Powell watershed suggest that agricultural and urban land uses contribute sediment to the stream, increasing embeddedness and reducing cover for fish and invertebrates (Figure 5-2 and 5-3). Individual habitat parameters, including bank vegetative protection, bank stability, and riparian vegetative protection, were positively correlated with forested land in 1- and 2-km-long corridors. Near-stream pasture and developed land, especially along small streams, were associated with degradation of overall habitat quality as well as instream habitat, bank stability and vegetation, and channel morphology. Land uses with biota comparisons indicated that up to 55% of the variability in fish IBI could be explained, with proximity to mining and urban land use having the most adverse effects (Table 5-1, Figure 5-5). Not only were all types of biota adversely affected by increasing amounts of mining and urban areas in relative proximity to the sampling site, but fish IBI scores actually improved at sites located near pasture lands. This was contrary to results found for the pilot study in the Copper Creek watershed (Figure 4-5), where proximity to pasture lands was associated with less mussel species richness. This occurred because the Copper Creek subwatershed is not heavily influenced by mining or urban areas. Thus, the Clinch and Powell analysis indicates that far more adverse effects on biota in this watershed occur from mining and urban areas than from pastureland. However, habitat degradation from near-stream pastureland would likely be a more obvious stressor if toxic mining effects were removed.

Because anthropogenic land use is prevalent throughout all regions of this watershed, other landscape characteristics, such as catchment slope, elevation, stream size, and site-specific factors, may act as determinants of mussel species richness and abundance in certain areas of the watershed. Sediment deposition and scouring in response to landscape-dependent flow variations throughout the watershed may result in differences in habitat quality and, consequently, differences in mussel fauna. There is some indication that high-slope catchment areas require longer riparian corridors to ensure mussel species richness, whereas shorter riparian areas are adequate in areas with lower catchment slopes.

It was difficult to assess the influence of different-sized riparian zones and the land uses within riparian corridors on fish assemblage integrity because of the confounding effects of mining activities and the limited amount of data. However, our analyses confirm the conclusions of previous analyses in Chapter 5 that describe mining and urban land as being the major factors affecting fish assemblages (i.e., stressors from these sources are the limiting factor to fish assemblage integrity). Habitat quality, which was shown to affect IBI scores (Section 5.4), appeared to be influenced more by riparian land use than by whole-drainage-area land use. Furthermore, habitat quality in small, high-elevation, and high-gradient streams seemed to be more influenced by riparian conditions than did the habitat in larger, lower-elevation, and lower-slope streams. Additionally, these results support those presented in section 5.5 that describe the detrimental effects of multiple stressors on fish assemblages.

Although mining and urban land are the only landscape factors that can be implicated as stressor sources using current data, it is likely that other land-use features are also detrimental to fish assemblages. Analysis of riparian land use indicates that streamside pasture and possibly other human land uses were detrimental to habitat quality; however, because of the toxic effects from mines (which are more often located in forested areas and absent from pasture lands), land-use effects on fish via habitat degradation (e.g., sedimentation due to pasture runoff) were not evident. Therefore, although reducing stressors from mining operations is likely to improve overall fish IBI scores, it is not clear that this will restore IBI scores to relatively unimpaired reference sites in the watershed. Once mining-related stressors are reduced, land-use-related habitat degradation may become the limiting factor to fish assemblage integrity.

6.1.3. Unexplained Variance

All of our risk analyses indicated that nearly half of the variance in biological measures of effect was still unexplained, given the land cover and habitat data available. More detailed analyses of riparian corridor land uses and physical attributes of catchments (e.g., slope, elevation) improved the relationships between sources and measures of effects (Section 5.6).

However, our analyses indicate that other factors could have impacted IBI and mussels, including

- Wastewater discharges and other point and nonpoint sources that could release toxic constituents downstream (Lingenfelser, 2000),

- Episodic toxic spills (Jones et al., 2000),

- Habitat fragmentation, and

- Lack of sufficient fish hosts at the necessary spawning times (Sheehan et al., 1989).

Results of our analyses suggest that these types of site-specific factors may be important in explaining variability in fish and mussel abundance and distribution. Although not explicitly included in our analyses because of data limitations, mussel species richness data collected over 93 years show sharp declines in native mussels following spills of toxic materials (Figure 5-8; Sheehan et al., 1989). Several toxic spills have occurred in this basin over the past 30 years, including a 1999 truck accident that spilled concentrated ammonia into Cedar Creek in the upper Clinch River, resulting in a large fish kill and mortality of at least 300 federally threatened or endangered mussels (Jones et al., 2000). Mussels have still not recovered from these spills, possibly because of residual sediment contamination (Van Hassel and Gaulke, 1986), which may impair survival of mussel glochidia and larvae (Kauss and Hamdy, 1991).

The lower amount of variance explained by land use for the EPT (Tables 5-1 and 5-2), as compared to the fish IBI, could be due in part to the coarse taxonomy used for invertebrates (family level) and potential loss of information (Barbour et al., 1999). However, we cannot rule out the fact that invertebrates have relatively short life cycles and may be able to recolonize or recruit individuals quickly following stress. Several researchers, including Cairns et al. (1971), Minshall et al. (1983), and, in this watershed, Crossman et al. (1973) have noted relatively rapid recolonization of macroinvertebrates following episodic events. However, as native mussels and fish have yet to recolonize this area of stream, EPT may not be as sensitive an indicator of past water quality effects as either native mussels or fish.

The even lower amount of variance explained for mussels species richness in our analyses (Table 5-3) could be due to a number of other factors, including

- Site-specific geomorphic characteristics such as substrate particle size, flow and current velocity, and orientation of bedrock ridges (Church, 1996),

- Proximity to episodic spills that could not be adequately analyzed in this risk assessment, and

- Variance from year to year in fish host assemblage in the area.

6.1.4. Stressor-Response Relationships

Relationships between habitat quality and biological measures of effect showed that biota were influenced positively by instream cover and negatively by embeddedness. Given that we observed negative relationships between pastureland cover and riparian integrity and proximity to urban lands and embeddedness, it is not surprising that these types of land use affect fish assemblage integrity. This idea is further supported by analyses presented in Section 5.6 that show positive correlations between near-stream anthropogenic land uses (i.e., pasture and developed land) and habitat degradation, particularly in smaller, higher-elevation, higher-sloped streams.

The relationship between the cumulative number of stressors at a site and mussel species richness or fish IBI (Figure 5-17) suggests that fish and native mussel populations are relatively vulnerable to risks in this watershed. The more stressors present, the more likely further extinctions or extirpations will take place unless additional resource protection measures are taken.

Several lines of evidence described above (and summarized in Table 6-1) point to the importance of various land-use activities and the riparian corridor integrity as determinants of native mussel and fish distribution in the Clinch and Powell watershed. Lines of evidence include analysis of field data collected by TVA and other organizations, as well as information from published studies in other watersheds. Key factors appear to be sedimentation and other forms of habitat degradation from urban and agricultural areas, as well as toxic chemicals from coal mining operations and urban areas.

The importance of riparian zone characteristics on instream habitat quality and aquatic fauna observed in this study has been reported in many other lotic systems (Minshall et al., 1983; Cooper et al., 1987; Gregory et al., 1991). This study further clarified that the strongest relationships between forested riparian areas in Copper Creek and biological and habitat measures occurred with a riparian width of 200 m and 500 to 1,000 m upstream of the sampling site. Areas within these limits that had predominantly forested land cover tended to have less

Table 6-1. Summary of stressors affecting assessment endpoints in the Clinch and Powell watershed risk assessment and lines of evidence used to characterize the risk and recovery potential from stressors

Type of fauna	Life stage	Stressors	Lines of evidence	Risk	Recovery potential
Native mussels	Glochidia (larvae)	Sedimentation, toxic chemicals, lack of fish host, low flows/drought	Analyses of land use, spill effects, TVA habitat assessments, TVA CMCP data, and TVA CPRATS fish IBI data; literature review	Mortality, reduced growth	Poor unless recruiting areas are nearby; higher likelihood in nonheadwater areas
	Juveniles	Sedimentation, toxic chemicals, low flows/drought		Mortality, reduced growth	
	Adults	Sedimentation, toxic chemicals		Mortality, reduced growth and reproduction	
Native fish	Spawning	Sedimentation, low flow, high temperature	Literature	Reduced reproduction	Higher probability if natural riparian corridor vegetation sufficiently intact and >50 m on either side of the stream
	Juvenile growth	Toxic chemicals, sedimentation, lack of flow	Literature review; analyses of land-use and TVA habitat assessments	Poor growth, mortality	
	Adult	Toxic chemicals, lack of flow, high temperature	Literature review; analyses of land-use, habitat, and TVA CPRATS fish IBI data	Poor growth, mortality	

sedimentation, more instream cover for aquatic fauna, less substrate embeddedness, higher fish and native mussel species richness, and a higher number of threatened and endangered species than did riparian areas that had > 50% agricultural area (crop or pasture). However, a 1999 mussel survey of this stream showed a loss of mussel species when compared with a similar survey performed in the 1980s, apparently from more pervasive sedimentation in the stream and frequent wading in water by livestock (Don Gowan, TNC, at the Clinch Assessment Workgroup meeting, December 1, 1998). Thus, if agricultural or livestock use within the upstream riparian zone is great enough, sedimentation effects and subsequent loss of habitat will ensue for some distance downstream, depending on stream gradient, flow, and channel morphology, even though forested riparian areas may be present at a given site (Lenat, 1984; Richards and Host, 1994).

Analyses from Tazewell County (Section 5.6) indicate that riparian land uses can have varying effects on biota, depending on landscape factors such as slope, elevation, and stream size. Preservation or restoration of mussel communities in small, high-slope streams may require long zones (5–10 km) of riparian protection; shorter zones (1–2 km) of riparian protection may be required in larger, lower-gradient streams. Analysis of habitat data from stations throughout the Clinch River watershed also demonstrated the importance of riparian zones to stream integrity. Urban and pasture land use in riparian zones appeared to affect overall habitat quality as well as individual habitat components (including instream habitat, bank characteristics, and morphological features) more than did whole-drainage land use.

Although riparian vegetation can reduce deleterious land-use effects on water quality (Lowrance et al., 1984; Gregory et al., 1991; Osborne and Kovacic, 1993), it is not clear that, in this watershed, improvement of the riparian corridor alone in this watershed will result in recovery of native mussel and fish populations. Several researchers have reported significant effects of upland land uses on surface water quality, depending on the spatial pattern of those uses in the watershed (Omernik et al., 1981). In the Clinch and Powell watershed, there have been several reports of little or no recovery of threatened or endangered mussel or fish species, despite improved water quality (O'Bara et al., 1994; Dennis, 1985; Ahlstedt, 1991). Recent mussel introduction efforts (Sheehan et al., 1989) may improve mussel recruitment and population stability, but the lack of recovery thus far may be due to too few host fish in the area (Zale and Neves, 1982a, b; Watters, 1996, 1997) or residual sediment toxicity (Van Hassel and Gaulke, 1986; Sheehan et al., 1989). This assessment was unable to evaluate these factors, and there is a general lack of relevant information on such effects in the Clinch and Powell watershed and other systems.

Another suggested cause for the decline in mussels over the past 70 years is more frequent summer drought conditions and lower base flows in general throughout the watershed

(Ahlstedt and Tuberville, 1997). A study conducted by The Nature Conservancy (Richter, 1996) of various hydrologic measures at the only two long-term USGS gauge stations in the watershed (Cleveland and Spear's Ferry on the Clinch River) concluded that there were few significant trends over time. However, the study also reported lower August flows in recent decades and less "flashiness," or changes in hydrograph rises and falls. The latter measure may be due to greater reforestation of the watershed in recent years, but it could also be due to fewer prolonged or large precipitation events. Lower summer flows would be detrimental to mussels and native fish, particularly if riparian vegetation in the upper watershed and tributaries is removed, thus increasing water temperatures (Vannote et al., 1980; Morris and Corkum, 1996).

With any endemic population, there is a high risk of extirpation from habitat fragmentation, resulting in populations that are too inbred and small in size and that are more susceptible to stressors. Native mussels and fish in the Clinch and Powell watershed may be no exception. Populations are now more widely separated than they have been historically, which could lead to reduced recruitment success and declining populations. For this reason, it appears to be most useful to concentrate protection efforts on those populations that appear most vulnerable because of their proximity to mining, urban areas, or transportation corridors. Protection and/or enhancement of the riparian corridor at these sites, as well as protection from toxic spills and discharges, is as important to sustaining endemic species as stocking new or historically important areas. If stream habitat as well as water quality can be maintained or improved, present mussel and fish populations might be able to expand into nearby areas, thus increasing the distribution and abundance of these species.

6.2. SOURCES OF UNCERTAINTY

Two general types of uncertainty were encountered in this assessment: (1) uncertainty concerning the reliability of source, stressor, and biological data; and (2) uncertainty concerning extrapolation of results from one biological measure to another or from one subwatershed to another.

6.2.1. Reliability of Source, Stressor, and Biological Data

Direct stressor data for this risk assessment were fairly limited, both in terms of the quantity and types of data available. TVA's habitat quality assessment information, obtained in conjunction with fish and benthic macroinvertebrate data in the CPRATS, was the major source of physical stressor data. This information is highly qualitative, and it is an indirect measure of actual physical stressors. We assumed, for example, that high substrate embeddedness at a site was a reflection of a high loading of fine material from the surrounding land activities. There is,

however, an unknown degree of uncertainty associated with this assumption. Detailed physical descriptions of each sampling site were not available or evaluated in this risk assessment. It is possible that certain sites may have been natural deposition areas, for example (owing to gradient and channel morphology), which would have led to erroneous associations between surrounding land uses or riparian vegetation characteristics and instream habitat measures. Given TVA's sampling protocol for selecting sampling sites, this is probably a minor source of uncertainty in our analyses, but it is a source nonetheless.

A more serious source of uncertainty associated with TVA's habitat data is that they are a qualitative index and each habitat measure is rated on an ordinal scale of 1 to 4. In some analyses (e.g., multiple regressions) these measures were treated as continuous variables, which may have introduced unknown biases. Furthermore, we assumed that there was consistency in the way in which sites were characterized, that is, there was little or no subjective bias in how habitat measures were derived for each site. Although this assumption is likely to hold, given TVA's documented training and habitat assessment protocols, our experience suggests that some investigator bias cannot be avoided when using qualitative assessment protocols. One recommendation to resource managers is to continue conducting physical habitat assessments along with biological collection efforts but to consider using more robust habitat assessment techniques, such as the revised Rapid Bioassessment Protocol (Barbour et al., 1999).

The combined potential effect of the above sources of uncertainty are perhaps best illustrated in the habitat quality measures reported for the upper Powell River. Contrary to workgroup expectations and some published data, we were unable to identify a significant relationship between either stream embeddedness or sedimentation and proximity to mining or number of mines nearby. This apparent paradox could be explained, in part, by uncertainties related to the qualitative nature of the habitat measures (including the fact that only four different values are possible) and by uncertainties related to natural geomorphic differences among sites that may mask land use-habitat quality relationships.

A related source of uncertainty is the reliance on dual-threshold categories of stress rather than on a gradient of stress values in much of this risk assessment. Threshold characterizations are acknowledged to be somewhat approximate and empirically defined in this risk assessment. More complete spatial coverage of stressor data would enable us to more quantitatively analyze gradient stressor-response effects.

Aside from physical habitat stressors such as sedimentation, this risk assessment recognized the potential importance of chemical stressors in the watershed. Unfortunately, we were unable to characterize chemical stressors owing to a lack of relevant data. The entire watershed has only two long-term water quality stations, both of which are located in the lower

part of the watershed. Although trend analyses were performed on the available data by statisticians at Virginia Tech University, Blacksburg, VA (Zipper et al., 1991), nearly all of these analyses were for conventional pollutants (biochemical oxygen demand, pH, fecal coliform). Potential chemicals of concern such as pesticides, coal mining chemicals, and heavy metals were poorly represented in STORET and other databases because they were largely unmeasured. Thus, water quality stressors were largely inferred in this risk assessment, based on nearby land-use/source activities in association with biological effects and habitat quality information. An example of this inference process is the biological effects data presented for a hydraulic oil commonly used in coal mining and related effects of proximity to mines (where this oil is used) on mussel and fish abundance and distribution in the upper Powell River.

The biological data used in this risk assessment were also subject to some of the same uncertainties as the habitat quality information. Some sites may have had relatively poor faunal representation because of natural geomorphic features that would mask statistical relationships between biota measures and land use or habitat quality characteristics. For example, it has been demonstrated that native mussel abundance is related, in part, to the orientation of bedrock ridges, which is a consequence of the direction of stream flow, the local geology, and location with respect to the inside or outside bend of the stream. None of these parameters were included in TVA's habitat measures, although they could perhaps be modeled by using available geology, topography, and digital elevation information in the GIS. TVA's CMCP mussel data are perhaps less susceptible to this source of uncertainty than are fish or macroinvertebrate measures because trained experts chose mussel sampling locations on the basis of historical knowledge and an experienced understanding of preferred mussel habitat. However, trained experts do not always locate preferred habitat or the best reference sites. Therefore, it should be noted that the presence of trained experts does not always translate into lower uncertainty.

As noted in several of our risk analyses, the macroinvertebrate measure EPT was associated with a moderate degree of uncertainty because family-level taxonomy was used, resulting in a relatively narrow-ranging index throughout the watershed. One recommendation to resource managers is to consider using lower-level taxonomy (genus or, preferably, species) and developing a suite of sensitive reliable metrics that are demonstrated to respond to human activities. Fish IBI data were likely to have less associated uncertainty, because the metrics in this index have been demonstrated to be sensitive in a number of other watersheds. However, fish collection methods often have unknown or unquantified efficiency, resulting in uncertain reliability in fish abundance and distribution data. Unfortunately, fish collection efficiency is typically not uniform across different-sized streams or different habitat types. Therefore, there

might be a bias in some of the fish IBI data, resulting in potentially inappropriate comparisons across sites. The magnitude of this source of uncertainty is unknown.

6.2.2. Extrapolation of Results Between Biological Measures

The two assessment endpoints of interest in this risk assessment were concerned with native mussel and native fish species. Although substantial information has been collected throughout the watershed for both types of fauna, much of these data were not used in this risk assessment because (1) they were not easily accessible (i.e., TVA's CMCP mussel data were archived on TVA's mainframe computer, and documentation of data codes was lacking) or (2) they were not provided in time for this project schedule (e.g., the Virginia Heritage database for fish and mussels). As a result, we supplemented available mussel data with fish IBI and macroinvertebrate EPT measures in the hope that we could extrapolate source/stressor-effect relationships to native mussels and fish. However, as demonstrated in our risk analyses, there was a high degree of uncertainty associated with extrapolating EPT measures to native mussel data; EPT did not necessarily respond to sources or stressors in a similar manner as mussels. Fish IBI, however, was a reasonable surrogate indicator for native mussel species richness, although there is some uncertainty (albeit lower than for EPT) in the quantitative relationship between these two fauna.

The apparent relationship between fish IBI and mussel species richness or abundance could be explained in more detail than was possible in this risk assessment. IBI is composed of a number of metrics, one of which is native species richness. We were unable to obtain individual IBI metric values for all CPRATS sites, but these data do exist. With some further effort, these data could be obtained and compared with available CMCP mussel data. Such an analysis would also yield a direct measure relevant to the native fish assessment endpoint. However, any comparisons between native mussel and IBI or EPT data will be limited by the lack of overlap in sampling locations between CMCP and CPRATS data. As stated in our risk analyses, only eight sites in the entire watershed had mussel and IBI and/or EPT data. A recommendation to resource managers is to consider at least a pilot sampling program in which all fauna are sampled at each site (along with more robust habitat assessment measures), so that this source of uncertainty can be addressed and hopefully minimized.

7. MANAGEMENT IMPLICATIONS OF THE CLINCH AND POWELL VALLEY ASSESSMENT

The assessment process, in particular the development of the conceptual model and the performance of the mulitvariate analyses, furthered a better understanding of environmental problems. In assessing environmental risk, a number of Federal, State, and local environmental agencies and organizations came together to share data, explore and develop solutions, and undertake actions within the watershed. Risk assessment findings should also help direct the efforts of the newly formed Upper Tennessee River Basin Roundtable, which is composed of various individuals, agencies, and organizations that have an interest in protecting the watershed. Results should be useful to the roundtable as it begins comprehensive strategic planning for watershed protection. Additionally, the numerous watershed coalition groups within the basin can use the findings of the risk assessment to direct their efforts to protect and improve water quality within their watershed.

Pending the results of this assessment, the workgroup agreed to consider implementing several management objectives to maintain or restore the threatened, endangered, or rare native freshwater mussels and fish in the Clinch and Powell watershed. These management objectives are:

- Create and maintain vegetated riparian zones in urban, agricultural, industrial, and other developed areas to reduce nonpoint-source pollution and enhance habitat.

- Implement BMPs, such as minimum till and treatment of feedlot waste, to reduce nonpoint-source pollution.

- Contain and treat runoff from mining activities to reduce pollutant load and sedimentation.

- Install or improve sewage treatment facilities to reduce inputs of pollutants and nutrients.

- Adequately treat industrial discharges to reduce input of toxic pollutants.

- Create and maintain stormwater retardation and holding facilities for highways and developed areas to reduce sedimentation and runoff.

Risk assessment participants developed and improved understanding of the interrelationships between various components of the ecosystem and the manner in which human activities contribute to environmental problems within the watershed. The process of risk assessment helped lend further credence to what many professional resource managers had long conjectured about problems within the watershed, thereby providing more scientific support for taking actions to address problems. For example, there is now a better understanding of the contribution of sediment to the river from cattle grazing. Such risk assessment findings will be useful to USFWS and TVA personnel, who can now share this information with farmers and encourage them to take actions, such as building fences to keep cattle out of streams.

Key findings from the Copper Creek pilot study include the following:

- Optimal benefits to fish, mussels, and perhaps other invertebrates would be realized by maintaining the riparian corridor for a minimum of 500–1,500 m upstream and 100 m to either side of the stream for the site of interest. This riparian area could constitute a stream-specific, optimal riparian management area within which to better prioritize protection efforts.

- Local riparian mitigation techniques (< 100 m upstream of site) might not be as effective in enhancing fish or mussel diversity as somewhat larger riparian mitigation efforts. Local instream habitat characteristics may not be related to upland land uses if there is a wide vegetated riparian corridor in those areas.

Key findings from the analyses of the entire watershed include the following:

- Longer riparian corridor lengths (2–5 km) may be more appropriate in higher-gradient streams to predict effects of land uses on fish and mussels.

- Shorter riparian corridor lengths (1 km) may be appropriate in low-gradient reaches (e.g., some parts of the mainstem Clinch and Powell rivers) to protect biota from deleterious land use effects.

- Mine effluents and spills appear to have the greatest overall effect on mussels and fish, as compared to other human-activity sources in the watershed.

- Urban land use had significant effects on stream habitat, mussels, aquatic insects, and fish. Nonpoint-source runoff as well as wastewater treatment effluents may be responsible for these effects.

- Accidental chemical spills have had drastic effects on native mussel populations in several parts of the watershed. These spills are primarily associated with major transportation corridors (U.S. highways and railroad tracks) and large industrial facilities.

- Pasture and other agricultural activities were often associated with impaired stream habitat.

- Because of the strong inverse relationship between mining activities and biota in this watershed, other land use effects on stream habitat and biota were difficult to determine. Once mining stressors are addressed, native mussel and fish populations may improve to a point; then land use-related habitat degradation may be a limiting factor for these fauna.

Examples of management actions that will be considered by the USFWS and TNC on the basis of the overall risk assessment findings include

- Restoring additional abandoned mine lands throughout the watershed.

- Studying further the chemical makeup of discharges from coal mining and processing facilities and the toxicity of these discharges to aquatic species;

- Increasing the extent of forested riparian areas adjacent to and upstream of critical aquatic habitat sites for mussels and fish;

- Implementing better spill control mechanisms on roadways and railroads near sensitive streams and more spill contingency plans for the watershed, which will enable the Virginia Department of Transportation and other agencies involved in constructing highway projects on or near waterways to design those projects to reduce catastrophic events and minimize impacts of accidental spills;

- Studying further the impacts of urban development on aquatic species and working with planning and development agencies to identify and implement appropriate measures to protect aquatic resources;

- Improving monitoring and control of allowable limits of constituents from coal mining operations;

- Restricting the type of materials transported over certain bridges; and

- Installing BMPs for pasture and agricultural land to reduce sediment loading and implementing better treatment of wastewater discharges.

There are costs associated with implementing management decisions, and trade-offs must be made. As part of a follow-on study, the University of Tennessee obtained grant funding to determine how Clinch Valley residents would evaluate trade-offs involving environmental quality and economic factors in the Clinch Valley. The outcomes of the risk characterization were directly used in the survey design. Residents were asked to state their preferences between hypothetical management options that provided differing levels of quality of aquatic life, sportfishing, songbirds and other wildlife and that also had differing impacts on regional income. The risk characterization had demonstrated relationships between (1) degree of riparian agriculture and fish IBI and (2) proximity to coal mining operations and fish IBI. Therefore, in the survey, one set of hypothetical management options evaluated consisted of agriculture-free riparian zones of varying widths, and another set included changes in coal sector income that would be implicit in policies to de-emphasize that sector (Kahn et al., 2001). Survey results, when analyzed, will enable decisionmakers to score specific options as to their likely acceptance by the community, including individuals' willingness to be taxed or to accept compensation as part of implementing a given management strategy.

The watershed ecological risk assessment compiled and organized information from several sources into a usable data set, which is available from NCEA-W. This will benefit environmental managers as various agencies and organizations can more easily add to and use the data to further assess problems for other decision making purposes. For instance, the Southern Appalachian Man and the Biosphere (SAMAB) program will benefit from incorporating the recently developed data set into its database. The data and findings will be used by FWS to undertake environmental review of various federally funded and/or permitted projects, such as those considered under the authority of the Section 404 of the Clean Water Act

and the Surface Mining Control and Reclamation Act. The data set will be available to agencies and organizations such as FWS and TNC as they strive to develop plans and make decisions regarding actions to further the recovery of endangered and rare species.

Private landowners and natural resource managers in industry can use the findings to minimize and avoid impacts of activities on rare species and other fish and wildlife and also to develop habitat conservation plans for these species. Information developed through the assessment may also aid managers and conservation groups in efforts to obtain grants and assistance from various State- and federally sponsored programs. The vast majority of actions to remedy environmental problems within the watershed are likely to be accomplished without any direct regulatory actions and should benefit local economies and environments.

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APPENDIX A

STAKEHOLDERS INVOLVED IN RISK ASSESSMENT PLANNING

Table A-1. Agencies and conservation organizations active in the Clinch and Powell watershed

Federal government	TVA, Clinch River Action Team U.S. Environmental Protection Agency Department of Interior U.S. Fish and Wildlife Service U.S. Geological Survey National Biological Service Office of Surface Mining	Department of Agriculture U.S. Forest Service Natural Resource Conservation Service Consolidated Farm Services Agency Rural Economic and Community Development
State government	**VIRGINIA** Department of Conservation and Recreation Division of Natural Heritage Division of Soil and Water Conservation Division of Parks and Recreation Department of Agriculture and Consumer Services Soil and Water Conservation Districts Department of Forestry Department of Game and Inland Fisheries Department of Mines, Minerals, and Energy Division of Mined Land Reclamation (and other divisions) Department of Environmental Quality Virginia Cave Board	**TENNESSEE** Tennessee Wildlife Resources Agency Department of Environment and Conservation Division of Natural Heritage Division of Water Pollution Control Division of Abandoned Mine Land Reclamation Department of Agriculture Division of Forestry Division of Plant Sciences Soil and Water Conservation Districts Department of Housing and Urban Development Planning District Commissions
Organizations	**VIRGINIA** The Nature Conservancy Black Diamond Resource Conservation and Development Council Coalition for Jobs and the Environment Clinch/Powell Sustainable Development Initiative Southern Environmental Law Center Sierra Club Audubon Naturalist Society	**TENNESSEE** The Nature Conservancy Clinch-Powell Resource Conservation and Development Council Citizens for Wilderness Planning Save Our Cumberland Mountains Tennessee Ornithological Society Tennessee Scenic Rivers Association Friends of the Clinch and Powell Rivers Sierra Club
Universities and colleges	Virginia Polytechnic Institute and State University Tennessee Technological University University of Tennessee East Tennessee State University Tusculum College	Virginia Highlands Community College Southwestern Virginia Community College Empire Community College Clinch Valley College University of Virginia

APPENDIX B

NATIVE MUSSEL AND FISH SPECIES OF CONCERN IN THE CLINCH AND POWELL WATERSHED

Table B-1. Mussel species in the Clinch and Powell watershed

UC = Upper Clinch; CC = Copper Creek; LR = Little River; PR = Powell River; Hist = Historical; Ext = Extant

Species (** = Cumberlandian)	UC Hist	UC Ext	CC Ext	LR Ext	PR Hist	PR Ext
Actinonaias ligamentina	X	X			X	X
Actinonaias pectorosa **	X	X	X	X	X	X
Alasmidonta marginata	X	X			X	X
Alasmidonta viridis	X	X	X		X	X
Amblema plicata	X	X	X		X	X
Cumberlandia monodonta	X	X				X
Cyclonaias tuberculata	X	X				X
Cyprogenia stegaria	X	X			X	X
Dromus dromas **	X	X			X	X
Elliptio crassidens	X	X			X	X
Elliptio dilatata	X	X	X	X	X	X
Epioblasma arcaeformis**	X	X				
Epioblasma biemarginata	X	X				
Epioblasma brevidens **	X	X			X	X
Epioblasma capsaeformis **	X	X	X		X	X
Epioblasma florentina walkeri**	X	X				
Epioblasma haysiana**	X	X			X	X
Epioblasma lenoir**	X	X				
Epioblasma lewisi**	X	X			X	X
Epioblasma tortulosa gubernaculum**	X	X			X	X
Epioblasma triquetra	X	X			X	X
Fusconaia barnesiana**	X	X	X	X	X	X
Fusconaia cor **	X	X	X		X	X
Fusconaia cuneolus **	X	X	X	X	X	X
Fusconaia subrotunda	X	X			X	X
Hemistena lata	X	X			X	X
Lampsilis abrupta		X				
Lampsilis fasciola	X	X	X	X	X	X
Lampsilis ovata	X	X	X		X	X
Lampsilis ovata ventricosa	X	X			X	X
Lasmigona costata	X	X	X	X	X	X
Lasmigona holstonia	X	X			X	X
Lemiox rimosus **	X	X			X	X
Leptodea fragilis		X			X	X

Table B-1. Mussel species in the Clinch and Powell watershed (continued)

Species (** = Cumberlandian)	UC Hist	UC Ext	CC Ext	LR Ext	PR Hist	PR Ext
Lexingtonia dolabelloides**	X	X			X	X
Ligumia recta	X	X			X	X
Medionidus conradicus **	X	X	X	X	X	X
Pegias fabula**	X	X			X	X
Plethobasus cyphyus	X	X			X	X
Pleurobema coccineum	X	X				
Pleurobema cordatum	X	X				
Pleurobema oviforme**	X	X	X	X	X	X
Pleurobema plenum		X				
Pleurobema rubrum		X				
Potamilus alatus	X	X			X	X
Ptychobranchus fasciolaris	X	X	X	X	X	X
Ptychobranchus subtentum **	X	X	X	X	X	X
Quadrula cylindrica		X				X
Quadrula cylindrica cylindrica	X	X			X	X
Quadrula cylindrica strigillata	X	X	X		X	X
Quadrula intermedia**	X	X			X	X
Quadrula pustulosa pustulosa	X					
Quadrula sparsa**	X	X			X	X
Strophitus undulatus	X	X			X	X
Toxolasma lividus**	X	X			X	X
Truncilla truncata	X	X				X
Villosa fabalis	X	X			X	X
Villosa iris	X	X	X	X	X	X
Villosa perpurpurea **	X	X	X		X	X
Villosa trabalis**	X	X				
Villosa vanuxemensis vanuxemensis **	X	X	X		X	X
TOTAL:	X	X	X	X	X	X

Source: Steve Ahlstedt, U.S. Geological Survey, presented at a Clinch and Powell Workgroup Meeting, 1997.

Table B-2. Fish species in the Clinch and Powell watershed

H = Historical record

Species (** = Introduced)	Upper Clinch 06010205	Copper Creek 06010205	Guest River 06010205	Powell River 06010206
Ambloplites rupestris	X	X	X	X
Ameiurus melas**	X	X		
Ameiurus natalis	X	X	X	X
Ammocrypta clara	X			X
Ammocrypta pellucida				X
Aplodinotus grunniens	X			X
Campostoma anomalum	X	X	X	X
Carassius auratus **	X	X		X
Carpiodes carpio	X			
Carpiodes cyprinus	X	X		X
Carpiodes velifer	X			
Catostomus commersoni	X	X	X	X
Clinostomus funduloides	X			
Cottus baileyi	X			
Cottus bairdi	X			
Cottus carolinae	X	X		X
Cottus sp (broadbanded sculpin)	X			
Ctenopharyngodon idella**	X			
Cycleptus elongatus	X			
Cyprinella galactura	X	X	X	X
Cyprinella monacha	X			X
Cyprinella spiloptera	X	X		X
Cyprinella whipplei	X			X
Cyprinus carpio **	X	X	X	X
Dorosoma cepedianum	X	X		X
Dorosoma petenense **	X	X		X
Ericymba buccata**	X			
Erimystax cahni	X			X
Erimystax dissimilis	X	X		X
Erimystax insignis	X	X		X
Esox masquinongy **	X			X
Etheostoma blennioides	X	X	X	X
Etheostoma caeruleum	X			X
Etheostoma camurum	X	X		X
Etheostoma cinereum	X			

Table B-2. Fish species in the Clinch and Powell watershed (continued)

Species (** = Introduced)	Upper Clinch 06010205	Copper Creek 06010205	Guest River 06010205	Powell River 06010206
Etheostoma flabellare	X	X		X
Etheostoma kennicotti	X			X
Etheostoma percnurum	X	X		
Etheostoma rufilineatum	X	X		X
Etheostoma simoterum	X	X		X
Etheostoma stigmaeum jessiae	X	X		X
Etheostoma swannanoa	X			X
Etheostoma tippecanoe	X	X		
Etheostoma vulneratum	X	X		X
Etheostoma zonale	X	X		X
Fundulus catenatus	X	X		X
Gambusia affinis **	X			
Hiodon tergisus	X			
Hybognathus hankinsoni				X
Hybopsis amblops	X	X		X
Hypentelium nigricans	X	X	X	X
Ichthyomyzon bdellium	X	X		X
Ichthyomyzon gagei	X	X		
Ichthyomyzon greeleyi	X	X		X
Ictalurus furcatus	X			
Ictalurus punctatus	X	X		X
Ictiobus bubalus	X			X
Ictiobus cyprinellus	X			
Ictiobus niger	X			
Labidesthes sicculus	X			X
Lagochila lacera	X			
Lampetra aepyptera	X			
Lampetra appendix	X			
Lepisosteus oculatus	X			
Lepisosteus osseus	X	X		X
Lepomis auritus **	X	X	X	X
Lepomis cyanellus	X	X	X	X
Lepomis gibbosus**	X		X	X
Lepomis gulosus	X			X
Lepomis macrochirus	X	X	X	X
Lepomis megalotis	X	X	X	X

Table B-2. Fish species in the Clinch and Powell watershed (continued)

Species (** = Introduced)	Upper Clinch 06010205	Copper Creek 06010205	Guest River 06010205	Powell River 06010206
Lepomis microlophus**	X			
Luxilus chrysocephalus	X	X	X	X
Luxilus coccogenis	X	X	X	X
Lythrurus ardens	X			
Lythrurus lirus	X	X		X
Macrhybopsis aestivalis	X			X
Micropterus dolomieu	X	X	X	X
Micropterus punctulatus	X	X		X
Micropterus salmoides	X	X	X	X
Morone chrysops **	X	X		X
Morone saxatilis **	X			
Moxostoma anisurum	X			X
Moxostoma carinatum	X	X		X
Moxostoma duquesnei	X	X		X
Moxostoma erythrurum	X	X		X
Moxostoma macrolepidotum	X			X
Nocomis micropogon	X	X	X	X
Notemigonus crysoleucas**	X			X
Notropis ariommus	X	X		X
Notropis atherinoides	X			X
Notropis buchanani	X			
Notropis leuciodus	X	X		X
Notropis photogenis	X	X		X
Notropis rubellus	X	X		X
Notropis rubricroceus	X			X
Notropis sp. (palezone shiner)	X			
Notropis sp. (sawfin shiner)	X	X		X
Notropis spectrunculus	X	X		X
Notropis telescopus	X	X		X
Notropis volucellus	X	X		X
Noturus eleutherus	X	X		X
Noturus flavipinnis	X	X		X
Noturus flavus	X	X		

Table B-2. Fish species in the Clinch and Powell watershed (continued)

Species (** = Introduced)	Upper Clinch 06010205	Copper Creek 06010205	Guest River 06010205	Powell River 06010206
Noturus stanauli	X			
Oncorhynchus mykiss **	X	X	X	X
Percina aurantiaca	X	X	X	X
Percina burtoni	X	X		
Percina caprodes	X	X		X
Percina copelandi	X			X
Percina evides	X	X		X
Percina macrocephala	X	X		X
Percina maculata	X			X
Percina sciera	X	X		X
Phenacobius crassilabrum	X			
Phenacobius uranops	X	X		X
Phoxinus erythrogaster	X			
Pimephales notatus	X	X	X	X
Pimephales promelas **	X	X		X
Pimephales vigilax	X			X
Polyodon spathula	X			X
Pomoxis annularis	X			X
Pomoxis nigromaculatus	X			X
Pylodictis olivaris	X	X		X
Rhinichthys atratulus	X	X	X	X
Rhinichthys cataractae	X	X		
Salmo trutta **	X	X		X
Salvelinus fontinalis **	X			X
Semotilus atromaculatus	X	X	X	X
Stizostedion canadense	X			X
Stizostedion vitreum	X			X
TOTAL:	X	X	X	X

Source: Adopted from Jenkins and Burkhead, 1994

APPENDIX C

RIPARIAN CORRIDOR LAND USE ANALYSES, UPPER CLINCH RIVER

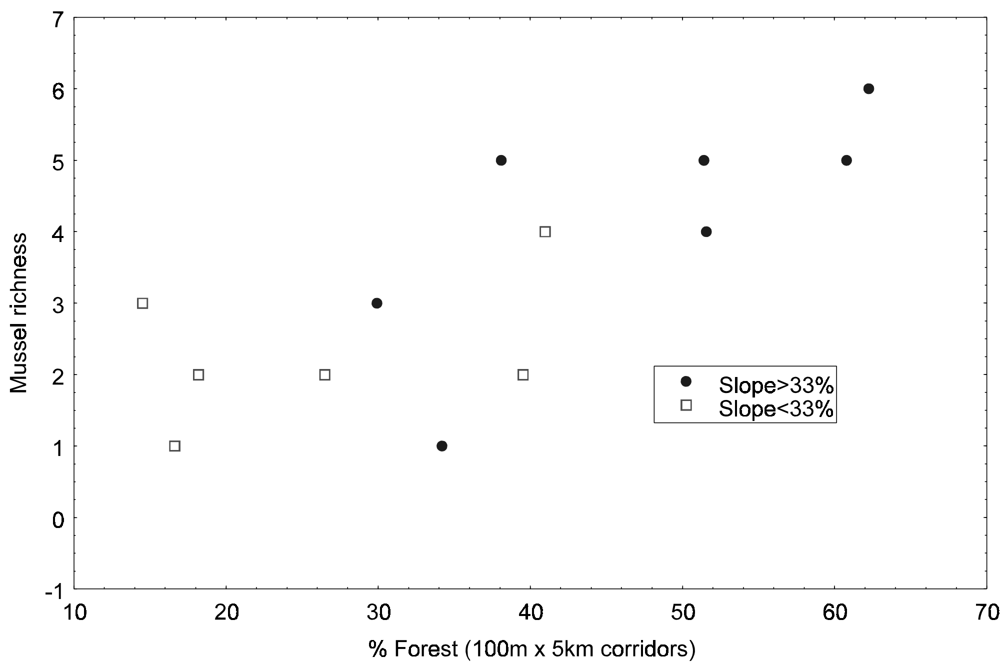

Figure C-1. Mussel richness versus the percentage of forested land within 100-m- wide, 5-km-long riparian corridors; streams are separated according to an average catchment slope threshold of 33%.

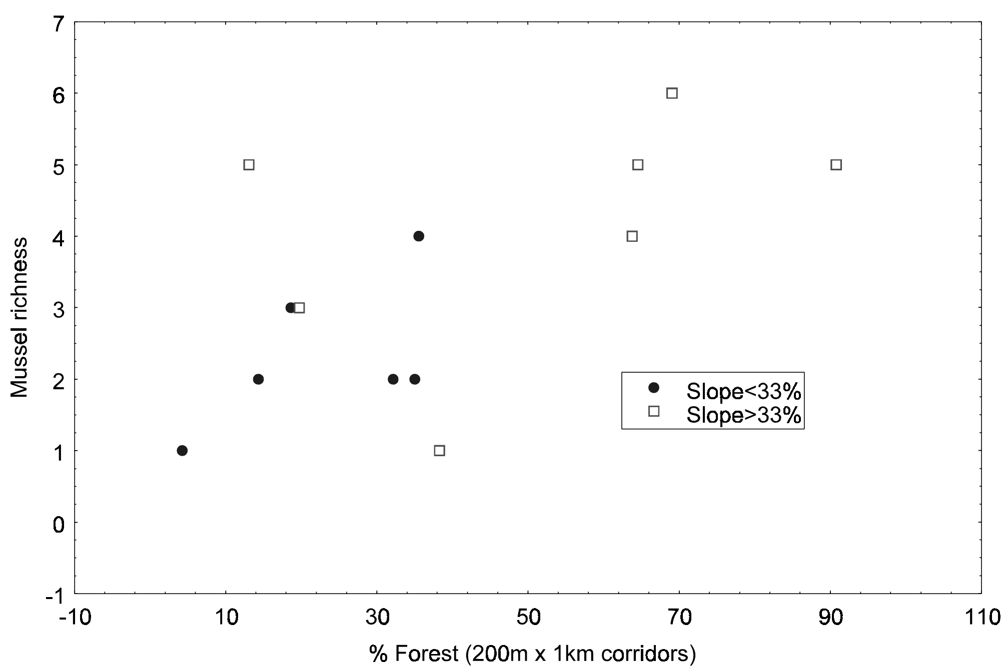

Figure C-2. Mussel richness versus the percentage of forested land within 200-m- wide, 1-km-long riparian corridors; streams are separated according to an average catchment slope threshold of 33%.

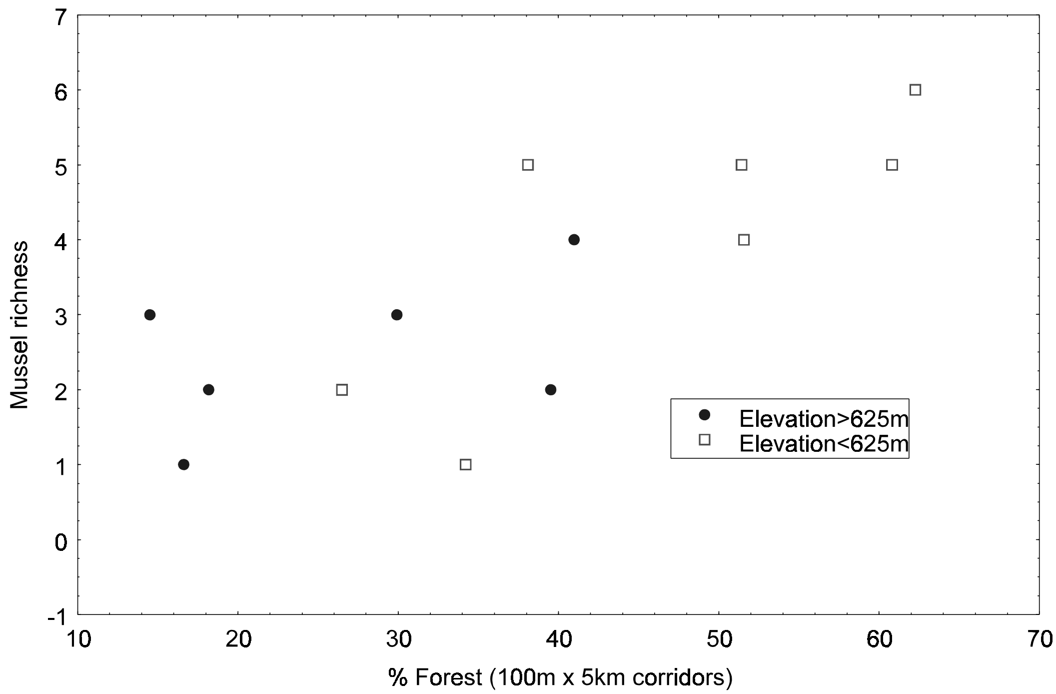

Figure C-3. Mussel richness versus forested land in 100 m × 5 km corridors with sites divided into site elevation categories (m).

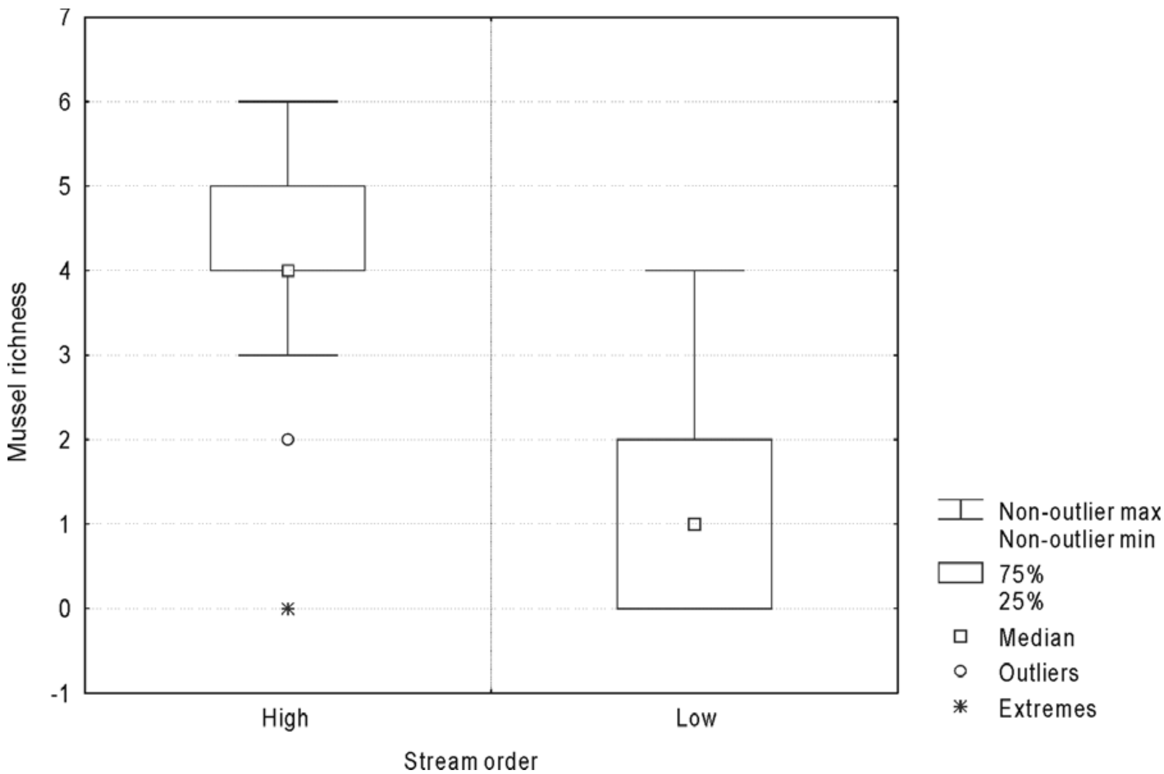

Figure C-4. Mussel richness in high (>4th order) and low (3-4th order) order streams.

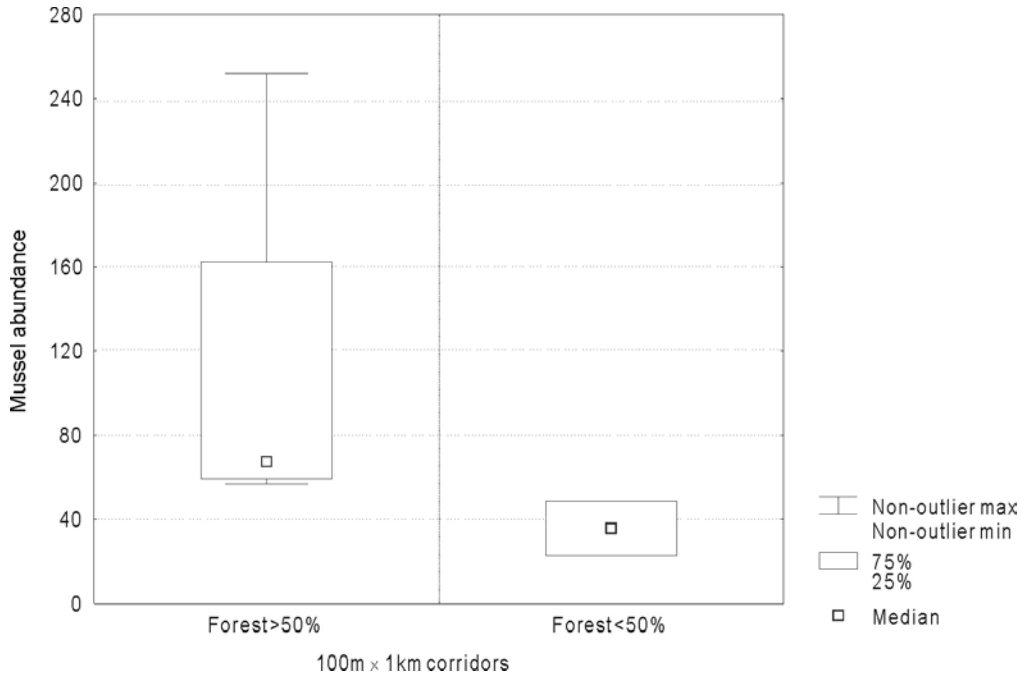

Figure C-5. Abundance of mussels in relation to forested land within 100-m-wide, 1-km-long riparian corridors of high-order streams (>4th order).

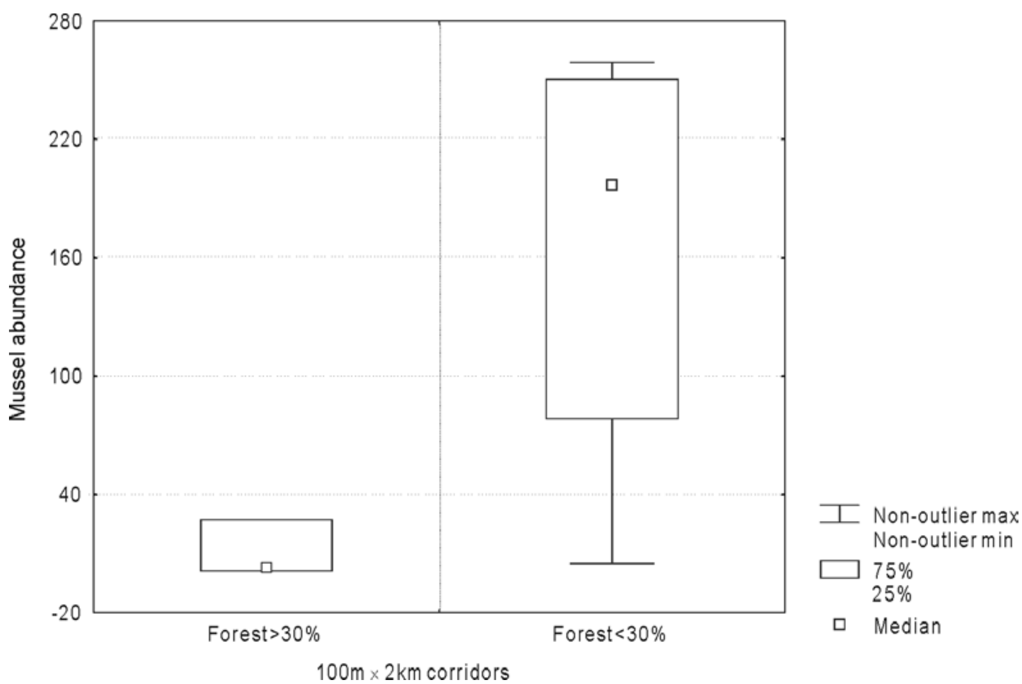

Figure C-6. Abundance of mussels in relation to forested land within 100-m -wide, 2-km-long riparian corridors of low-order streams (>4th order).

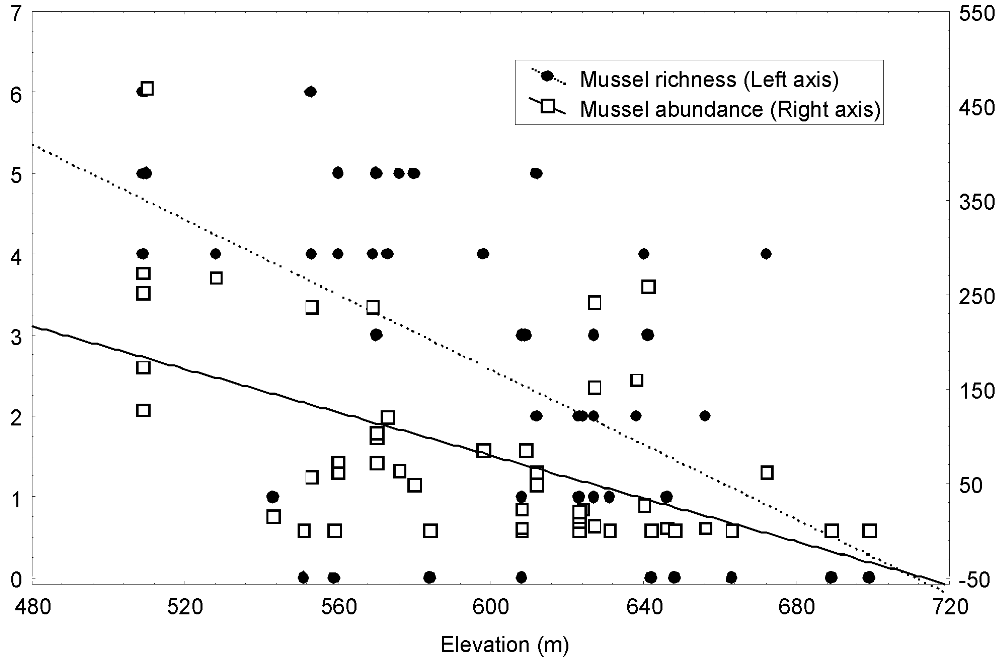

Figure C-7. Relation of mussel richness (left Y-axis) and abundance (right Y-axis) to site elevation (m) for high-order streams (>4th order).

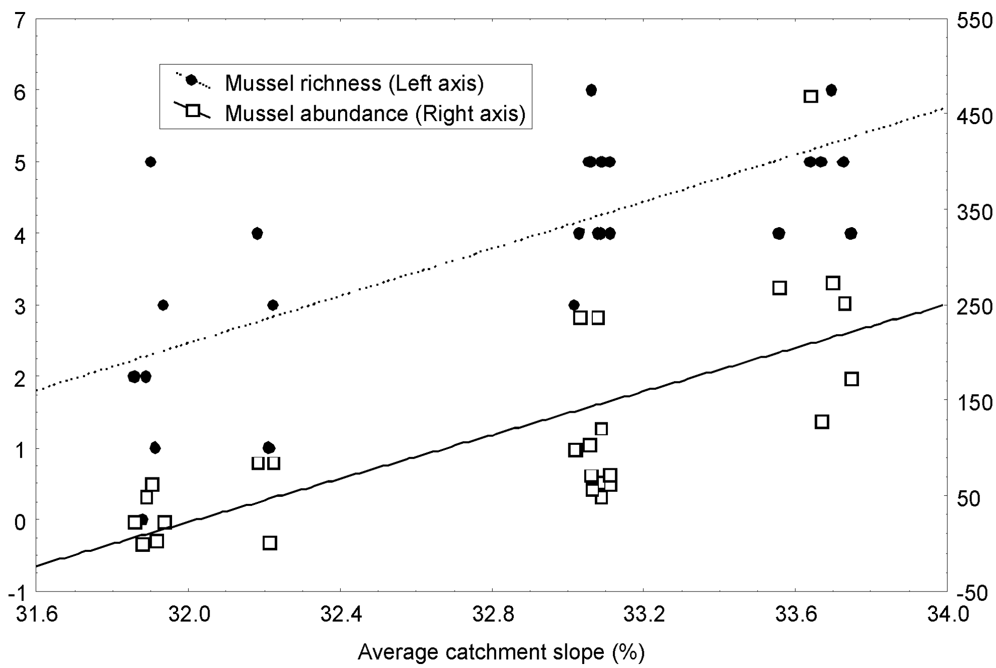

Figure C-8. Relation of mussel richness (left Y-axis) and abundance (right Y-axis) to average catchment slope (%) for high-order streams (>4th order).

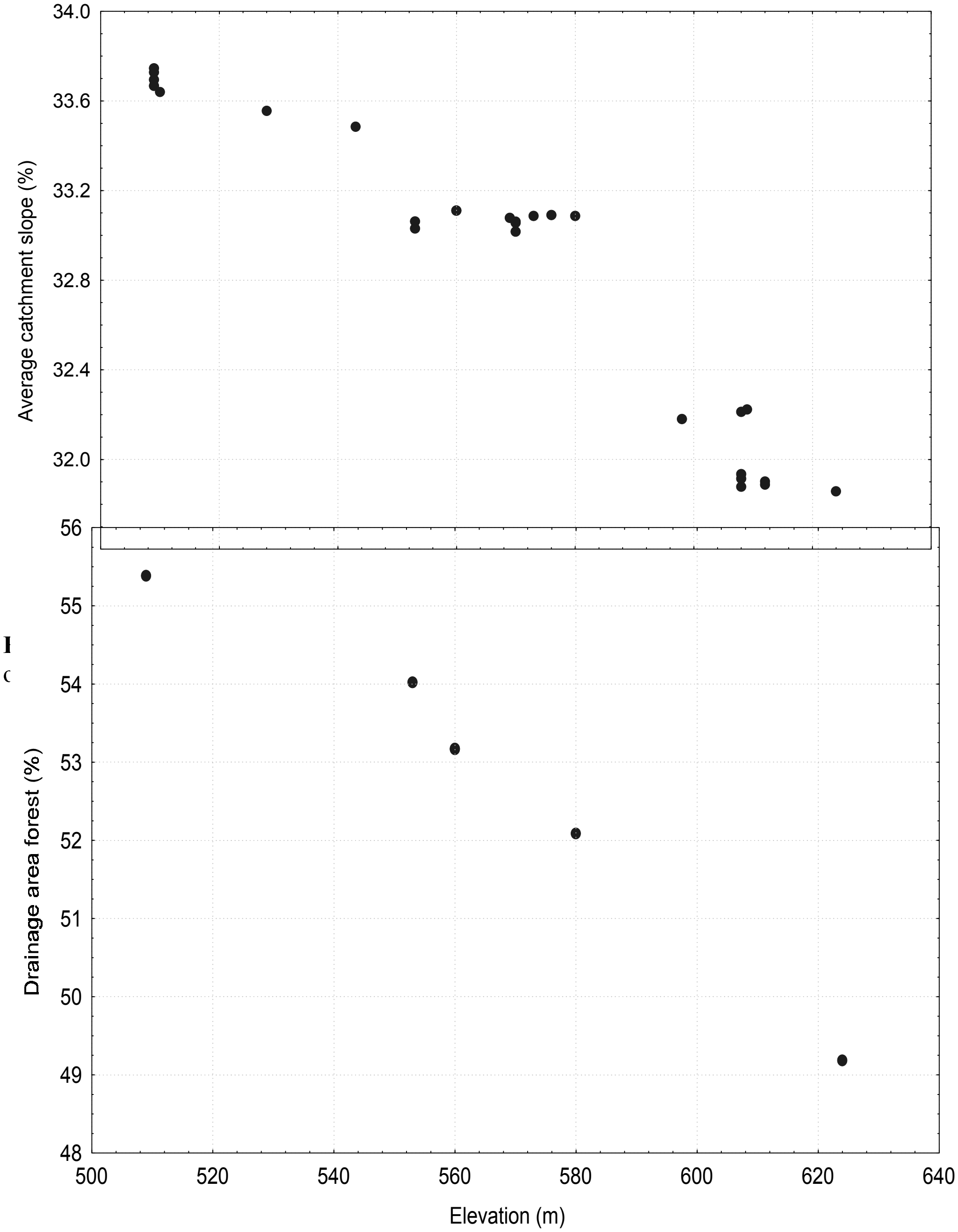

Figure C-10. Drainage area forested land (%) versus site elevation for high-order streams (>4th order).

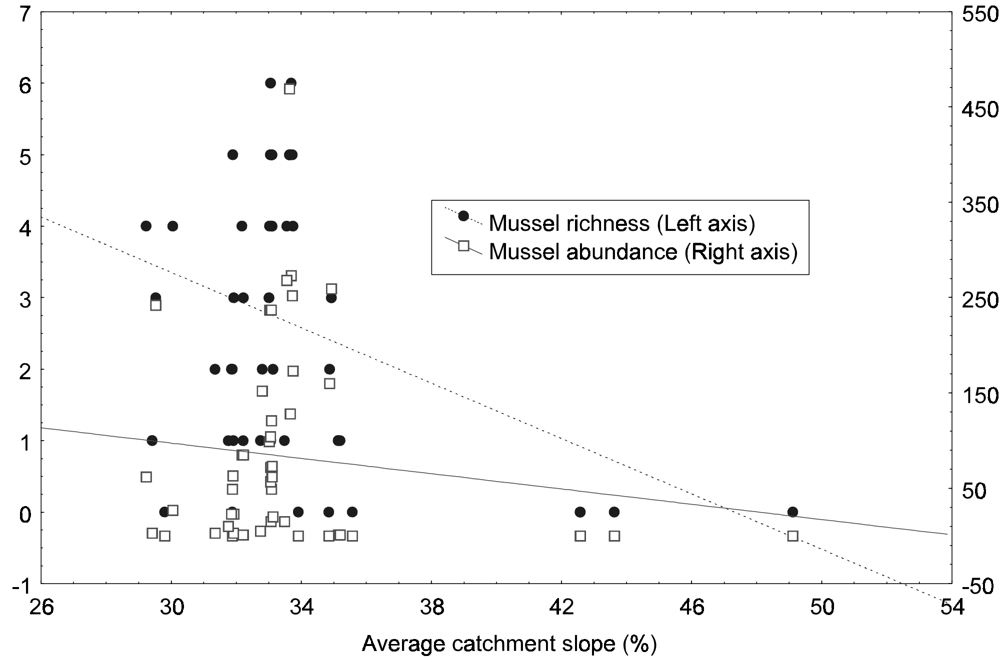

Figure C-11. Relation of mussel richness (left Y-axis) and abundance (right Y-axis) to average catchment slope (%) for low-order streams (3rd or 4th order).

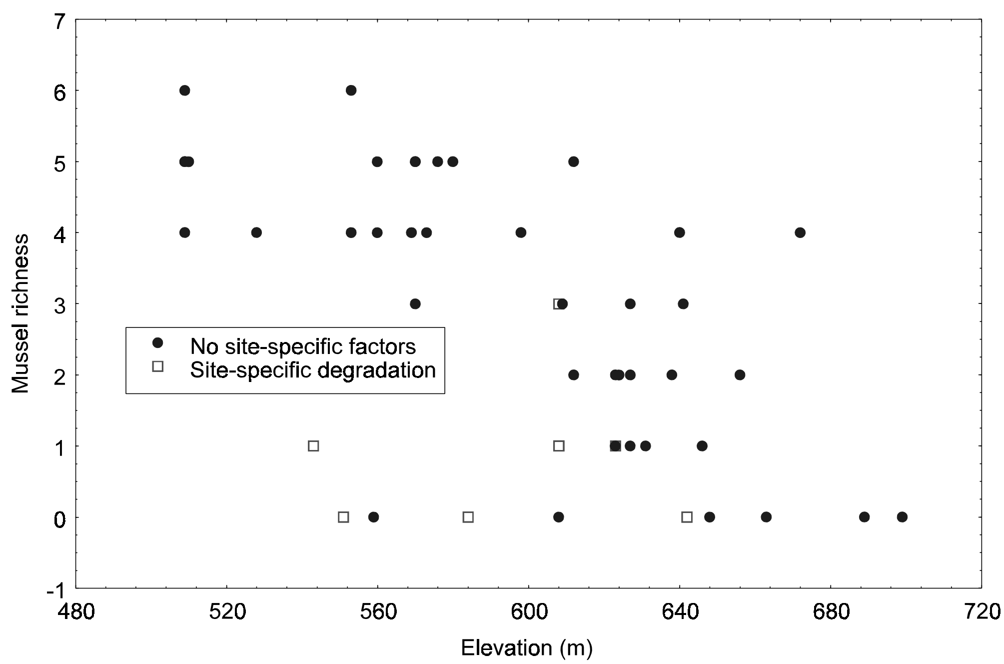

Figure C-12. Mussel richness versus site elevation for both sites with specific stressors and those with no site-specific stressors; site-specific degradation is seen here as outlier sites from broad-scale patterns.

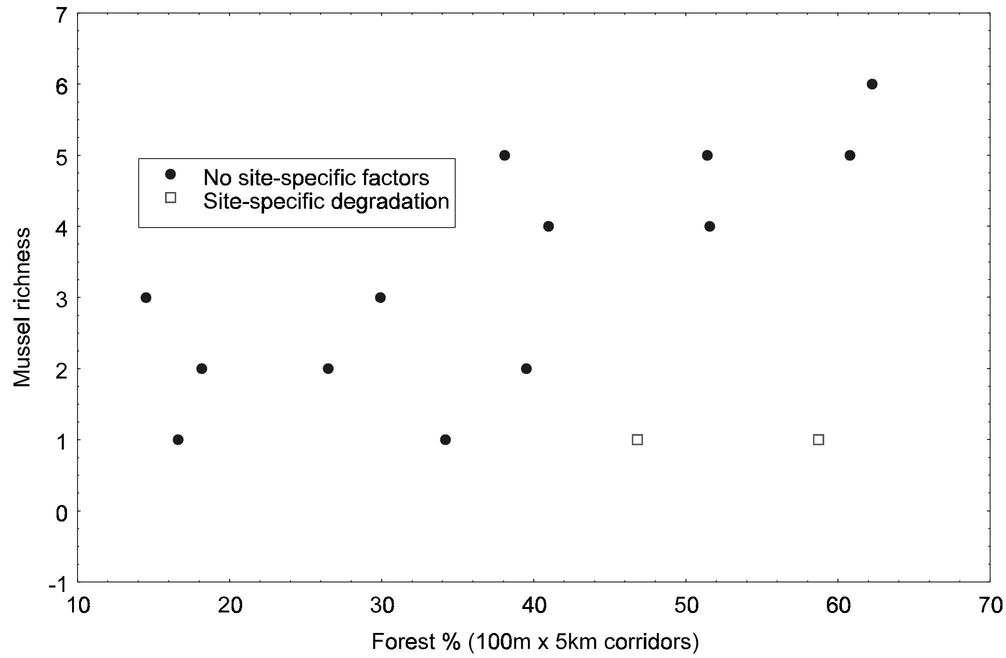

Figure C-13. Mussel richness versus corridor forested land for both sites with specific stressors and those with no site-specific stressors; site-specific degradation is seen here as outlier sites from broad-scale patterns.

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EPA/600/R-01/050
September 2002

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