

Abstract

POTTER, KEVIN MARK. Landscape characteristics and North Carolina stream life: A multiple-scale ecological risk assessment of nonpoint source pollution. (Under the direction of Frederick W. Cabbage).

Nonpoint sources of pollution may be responsible for as much as 50 percent of current water quality degradation in the United States, and as much as 70 percent in the Southeast. In this study, I used an ecological risk assessment methodology, at the watershed scale and riparian scales (zones 300, 100, and 50 feet on either side of streams), to analyze and quantify the impact of nonpoint pollution on the ecological integrity and water quality of North Carolina streams. Specifically, I determined how land-use patterns relate to aquatic ecological integrity, including the extent to which one of the most widely promoted best management practices (BMPs) – the preservation of riparian vegetated buffers – correlates with better ecological integrity.

The central goal of this project was the creation of a set of empirical models that describe the vulnerability of North Carolina aquatic ecological integrity – as measured by benthic macroinvertebrate community structure – to changes in the landscape-scale sources of nonpoint pollution. The models are the result of multiple regression analysis of Geographic Information System (GIS)-derived data, and take into account eight land form characteristics and three land cover types derived from 1992 Multi-Resolution Land Characterization (MRLC) Consortium raster data: forest, urban, and agriculture. The land form characteristics considered in this analysis are topographic complexity, mean elevation, watershed slope/relief ratio, watershed area, watershed shape, rainfall, soil clay content, and ecoregion.

The regression equation models created by this process can be used by managers and policymakers to weigh the risks of management and policy decisions for a given watershed or set of watersheds, including whether vegetated riparian buffers are ecologically effective and economically efficient in achieving water quality standards. The coefficient of multiple determination (R^2) for each equation indicates the proportion of variability in the invertebrate tolerance indices attributable to the landscape variables included in the model. The unstandardized regression coefficients for each landscape variable represent that variable's

weight and direction in the vulnerability index equation. The standardized (beta weight) regression coefficients indicate the relative importance of the landscape characteristic compared to the other landscape variables in the model equation.

The results of this study indicate that (1) landscape characteristics at the watershed scale predict variability in benthic macroinvertebrate community structure better than characteristics at the riparian scale; (2) land cover variables are of secondary importance to certain land form features, but are still significant predictors of macroinvertebrate community structure; (3) developed land use is the most important land cover variable at the watershed scale, while forested land cover is the most important at the riparian scale; (4) wider riparian buffer zones yield only minor differences in invertebrate community structure; and (5) more research is needed on how these interactions vary by the size of a watershed and the ecoregion in which it is located.

Based on these findings, it appears that water quality and stream ecological integrity may be most at risk in North Carolina watersheds where a greater amount of urban development is occurring at the watershed scale, where a lower percentage of forest cover exists in riparian corridors, and where the topography is generally flatter.

The ecological risk assessment process that produced these results was relatively simple and inexpensive. The results are straightforward and generally easy to interpret. The vulnerability model equations that resulted from this assessment process can provide a basis for quantitatively comparing, ranking, and prioritizing risks, which can be useful in cost-benefit and cost-effectiveness analyses of alternative management options. Specifically, the model equations offer a useful tool for characterizing the risk of potential land management options through the simulation of land use change, such as the conversion of land cover or the implementation of best management practices.

**Landscape Characteristics and North Carolina Stream Life:
A Multiple-Scale Ecological Risk Assessment
of Nonpoint Source Pollution**

by
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A thesis submitted to the Graduate Faculty of
North Carolina State University
in partial fulfillment of the
requirements for the degree of
Master of Science

Natural Resources

Raleigh

August 2002

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For my wife and my parents.

Thank you.

Biography

Kevin M. Potter is a native of Denver, Colo., where he was born in June 1971 and developed a deep appreciation for the natural landscapes of the Rocky Mountain region. After graduating from Regis Jesuit High School in Denver, he attended Drake University in Des Moines, Iowa, graduating magna cum laude in 1989 with a bachelor of arts in journalism and a minor in history. His honors thesis examined how the place names of Colorado's Rocky Mountains reflect the diverse and intriguing history of his home state. Since then, he has developed a manuscript, titled *Names Atop the Rockies: Chronicling Colorado's History Through the Naming of Its Mountains*, based on that research.

Potter spent the next four and a half years as a reporter at daily newspapers in Indianapolis, Indiana; Waterloo and Fort Dodge, Iowa; and Raleigh, North Carolina. The highlight of his journalistic career was two years and two months of 12-hour days working as political reporter for the 50,000-circulation *Waterloo/Cedar Falls Courier*. He covered Iowa's General Assembly and delegation to the U.S. Congress; reported on the 1996 Iowa presidential caucuses, the 1994 general election, and the 1996 primary and general elections; and wrote a weekly political analysis column. At *The News & Observer* in Raleigh, a newspaper with a circulation of 170,000, he worked the local police beat and wrote general assignment and enterprise stories, including articles about fire ants, carnivorous plants, unidentified flying objects, and poison-ivy.

From 1997 to 2001, Potter worked in the News Services office at North Carolina State University where, among other duties, he wrote and edited news releases about environmental- and natural-resource-related research, teaching, and extension efforts at NC State. He started work toward a master of science in natural resources, forest policy and management option, in 1998.

In Fall 2002 he begins work on a doctoral degree in forest ecology, with an emphasis on landscape ecology and conservation biology. His new research will focus on the conservation of genetic diversity in the fragmented Fraser fir populations of the Southern Appalachians.

Acknowledgements

As with any research project, this one would not have been possible without the generous support and patience of many people. I first want to thank my adviser and committee chair, Dr. Frederick W. Cabbage, who offered me the opportunity to pursue this research, and who has given me crucial guidance along the way. My other graduate committee members, Dr. Gary Blank and Dr. George Hess, have also offered me invaluable advice and assistance. Gary's thoughts were especially important in helping me develop the research project, while George helped me think through challenges in implementing the research plan and gave particularly useful comments on the draft of this manuscript.

I would also like to acknowledge the financial support I received from the U.S. Environmental Protection Agency through Science to Achieve Results (STAR) grant 2000-STAR-K3. Dr. Rex Schaberg has been a patient, generous, and understanding principal investigator for this grant; without his support, this project would not have been possible.

Dr. Marcia Gumpertz, Amy Nail, and Mark Atlas all gave me constructive and important statistics advice. As unofficial committee members, Atlas and Eileen Goldgeier have been helpful as the focus of my thesis research changed at least twice over the last two years. I appreciated and valued their support and patience.

Dr. Jim Gregory's instruction on watershed hydrology and stream ecology was essential for the successful completion of this project. The field trips to the Coweeta Hydrologic Lab near Franklin and to the Tidewater Research Center near Plymouth were particularly useful and enjoyable. David Lenat at the North Carolina Division of Water

Quality's Biological Assessment Unit offered critical insights about the use of benthic macroinvertebrate biotic indices in this research. I offer them my sincere thanks.

I also wish to thank the several people who assisted me with assembling and interpreting the Geographic Information System data used in this project: Steve Morris, James Jackson Sanborn, Bill Slocumb, and Andrew Bailey. Additionally, I appreciate the help from Dr. Bob Abt and Dr. SoEun Ahn regarding the USDA National Resource Inventory data. Most of the data used in this project were in the public domain, so thanks is also in order to the public agencies that made this information available and easily accessible: the U.S. Environmental Protection Agency's Office of Water, which offers thorough and useful data through the Better Assessment Science Integrating point and Nonpoint Sources (BASINS) program; the Multi-Resolution Land Characterization Consortium; the North Carolina Division of Water Quality's Biological Assessment Unit; and the North Carolina Center for Geographic Information and Analysis.

I offer my sincere thanks to my former colleagues in the North Carolina State University News Services office, who supported me even as my career trajectory changed dramatically.

Finally, I thank my parents, Mark and Diane Potter, whose love, dedication to education, and patience for a child's endless annoying questions long ago set my feet on the path I'm now traveling. Most of all, I thank my wife, Laura, who is my inspiration and who has supported and encouraged me, unconditionally, in more ways than I could count or list. You have made this possible, and for that I will always be grateful.

Contents

List of Tables	viii
List of Figures	x
Chapter 1: Nonpoint Source Pollution, Best Management Practices, and Ecological Risk Assessment	1
1.1 Policy Background	3
1.2 Forestry BMP Adoption and Compliance	8
1.3 Forestry BMP Effectiveness	9
1.4 Ecological Risk Assessment Overview	13
Chapter 2: Ecological Risk Assessment Methodology	16
2.1 Ecological Risk Assessment Strengths and Weaknesses	21
2.2 Project Overview	25
2.3 Ecological Risk Assessment Methodology	27
2.3.1 Phase 1: Problem Formulation	27
2.3.2 Phase 2: Risk Analysis	36
2.3.3 Phase 3: Risk Characterization	47
2.3.4 Phase 4: Communication of Assessment Results	65
Chapter 3: A Watershed-Scale Model for Predicting Nonpoint Pollution Risk in North Carolina	66
3.1 Abstract	67
3.2 Introduction	68
3.3 Methods	72
3.3.1 Landscape Variables	72
3.3.2 Macroinvertebrate Indices	73
3.3.3 Data Analyses	74
3.4 Results	76
3.4.1 Bivariate Regression Results	76
3.4.2 Multiple Regression Results	77
3.4.3 Reduced Regression Models	78
3.4.4 Ecoregion Results	79
3.5 Discussion	80
3.5.1 Risk Characterization	83
3.5.2 Uncertainty	87
3.6 Conclusions	88

Chapter 4: A Matter of Scale: The Best Landscape Predictors of Water Quality Depend on Watershed Size	100
4.1 Abstract	101
4.2 Introduction	102
4.3 Methods	108
4.3.1 <i>Landscape Variables</i>	108
4.3.2 <i>Macrobenthic Invertebrate Indices</i>	109
4.3.3 <i>Data Analyses</i>	110
4.4 Results	112
4.4.1 <i>Watershed-scale Analysis</i>	112
4.4.2 <i>Riparian-Scale Analysis</i>	112
4.4.3 <i>Importance of Topographic Complexity</i>	114
4.4.4 <i>Bivariate Regression Results</i>	114
4.5 Discussion	115
4.5.1 <i>Policy and Management Implications</i>	120
4.6 Acknowledgements	121
Chapter 5: Conclusions, Uncertainties, and Future Work	129
5.1 Conclusions	130
5.1.1 <i>Strengths and Limitations of Analysis</i>	133
5.2 Uncertainties	135
5.2.1 <i>Stochastic Variability</i>	136
5.2.2 <i>Knowledge Uncertainty</i>	138
5.3 Future Work	141
Bibliography	146

List of Tables

Chapter 1

Table 1.1: Annual average contribution of point and nonpoint sources of pollution to impaired river miles from 1988 to 1998 in the South	5
---	---

Chapter 2

Table 2.1: Variables used in analysis of stream invertebrate tolerance to stream degradation	43
---	----

Chapter 3

Table 3.1: Variables used in analysis of stream invertebrate tolerance to stream degradation	91
---	----

Table 3.2: Summary of relationship among landscape metrics, biotic indices, and water quality	93
--	----

Table 3.3: Results of simple regression analysis between land cover variables and both of the benthic macroinvertebrate tolerance indices	93
--	----

Table 3.4: Results of simple regression analysis between land form variables and two benthic macroinvertebrate tolerance indices	93
---	----

Table 3.5: Results of regression analysis for full landscape model and each of the two benthic macroinvertebrate tolerance indices	94
---	----

Table 3.6: Results of full-model regression analysis for the North Carolina Biotic Index	94
---	----

Table 3.7: Results of full-model regression analysis for the EPT Biotic Index	95
--	----

Table 3.8: Summary of stepwise regression results for two biotic index models	95
--	----

Table 3.9: Reduced model following stepwise regression for the North Carolina Biotic Index	95
---	----

Table 3.10: Reduced model following stepwise regression for EPT Biotic Index ...	96
---	----

Table 3.11: Summary of reduced models, following stepwise regression, for North Carolina's three ecoregions	96
--	----

Chapter 4

Table 4.1: Variables used in analysis of stream invertebrate tolerance to stream degradation	123
Table 4.2: Stepwise regression results for the North Carolina Biotic Index, watershed scale	125
Table 4.3: Stepwise regression results for EPT Biotic Index, watershed scale	125
Table 4.4: Stepwise regression results for the North Carolina Biotic Index, 100-foot buffer scale	125
Table 4.5: Stepwise regression results for the EPT Biotic Index, 100-foot buffer scale	126
Table 4.6: Comparison of simple regressions of land form characteristics at the watershed and riparian zone scales, versus biotic index scores	126
Table 4.7: Comparison of simple regressions of land-use characteristics at the 300-foot, 100-foot, and 50-foot riparian zone scales, versus biotic index scores	127
Table 4.8: Comparison of simple regressions forested percentage at the watershed scale and 100-foot buffer scale, versus biotic index scores, for the largest and smallest watersheds	127

Chapter 5

Table 5.1: Results of multiple regression analysis between land form variables and change in the North Carolina Biotic Index between 1992 and 1997	143
--	-----

List of Figures

Chapter 2

Figure 2.1: The risk assessment framework	19
Figure 2.2: The interrelationship of nonpoint source pollution, best management practices (BMPs), and aquatic ecological integrity	34
Figure 2.3: The relationship among landscape characteristics and sources of stress on the endpoint of the ecological risk assessment	40
Figure 2.4: 252 North Carolina Division of Water Quality benthic macroinvertebrate monitoring locations	50
Figure 2.5: An example of watershed delineation on the Oconaluftee River	51
Figure 2.6: The 74 North Carolina project watersheds	53
Figure 2.7: An example of 50-foot riparian buffer delineation	54
Figure 2.8: An example of 100-foot riparian buffer delineation	54
Figure 2.9: An example of 300-foot riparian buffer delineation	55
Figure 2.10: A close-up view of 50-foot, 100-foot, and 300-foot riparian buffers on the Watauga River	55
Figure 2.11: 1992 Multi-Resolution Land Characterization (MRLC) Consortium National Land Cover Data for North Carolina	57
Figure 2.12: Reclassification of 1992 Multi-Resolution Land Characterization (MRLC) National Land Cover Data for North Carolina	57
Figure 2.13: 1992 Multi-Resolution Land Characterization (MRLC) Consortium National Land Cover Data for North Carolina, aggregated into four classifications	58
Figure 2.14: The mean elevation and topographic complexity were found for each watershed	61
Figure 2.15: The mean elevation and topographic complexity were found for each buffer width	61

Figure 2.16: Precipitation data were taken from the rainfall monitoring stations nearest to the benthic sampling sites	63
Figure 2.17: Minimum soil clay content of the Upper Little River watershed	64
Figure 2.18: Minimum soil clay content of 300-foot riparian zones of the Upper Little River watershed	64
 Chapter 3	
Figure 3.1: The 74 North Carolina study watersheds	97
Figure 3.2: Reclassification of 1992 Multi-Resolution Land Characterization (MRLC) National Land Cover Data for North Carolina	98
Figure 3.3: 1992 Multi-Resolution Land Characterization land cover for the Swift Creek watershed	99
 Chapter 4	
Figure 4.1: The 74 North Carolina study watersheds	128
 Chapter 5	
Figure 5.1: U.S. Geological Survey eight-digit Hydrologic Units in North Carolina, and the 74 benthic macrobenthic monitoring sites	142

Chapter 1

Nonpoint Source Pollution, Best Management Practices, and Ecological Risk Assessment

Nonpoint sources of pollution – those that are not traceable to any discreet facility, but are induced by natural processes including precipitation (Ice 1988) – may be responsible for as much as 50 percent of current water quality degradation in the United States, and as much as 70 percent in the Southeast (West 2002). In this study, I used an ecological risk assessment methodology, at the watershed scale and riparian scales (zones 300, 100, and 50 feet on either side of streams), to analyze and quantify the impact of nonpoint pollution on the ecological integrity of North Carolina streams. Specifically, I determined how land-use patterns relate to aquatic ecological integrity, including the extent to which one of the most widely promoted best management practices (BMPs) – the preservation of riparian vegetated buffers – correlates with higher ecological integrity.

My goal was to create a set of empirical models that describe the vulnerability of North Carolina aquatic ecological integrity and water quality – as measured by macrobenthic invertebrate community structure – to the landscape-scale sources of nonpoint pollution. The models are the result of multiple regression analysis of Geographic Information System (GIS)-derived data, and take into account watershed land form characteristics and three land cover types: forest, urban, and agriculture.

Forest harvesting activities contribute to water quality degradation through the runoff of sediment, nutrients and other nonpoint pollutants; this situation is of particular concern in the Southeast, a region with extensive forestlands and high-quality aquatic resources (Chaplin *et al.* 2000). While compliance with voluntary and mandatory forestry BMPs is considered high throughout the Southeast (National Association of State Foresters 2001), it is unclear whether these measures effectively protect aquatic ecological integrity. Such

information is necessary to properly evaluate whether BMPs should be altered to achieve water quality standards in a cost-effective manner.

1.1 Policy Background

Before 1987, U.S. federal water quality law focused on the control of point source pollution, which emanates from closed systems including wastewater treatment plants and manufacturing facilities. As a result of the successful reduction of point source pollution, however, a greater proportion of water quality degradation is now associated with nonpoint sources, which may be responsible for as much as 50 percent of the nation's remaining water quality problems (Copeland 2001, Neary *et al.* 1988). In North Carolina, nonpoint source pollution is estimated to contribute to the degradation of 64 percent of degraded streams (N.C. Division of Water Quality 2000i). For the Southeast, that number is thought to be 70 percent (West 2002).

In 1987 the U.S. Congress shifted the focus of the Federal Pollution Control Act (widely known as the Clean Water Act) from controlling point source to nonpoint source pollutants, with the addition of Section 319 to the law. That amendment to the 1977 statute requires states to (1) identify water bodies not meeting water quality standards because of nonpoint pollutants, (2) isolate the nonpoint sources responsible, and (3) implement regulatory or voluntary programs to control them. Congress directed states to target resources for the most heavily impacted waters, develop management plans on a watershed basis, and identify BMPs to mitigate nonpoint source pollution impacts (Copeland 2001, Brown *et al.* 1993, Cabbage *et al.* 1993, Hohenstein 1987).

Additionally, in an attempt to restore impaired waters, the U.S. Environmental Protection Agency has proposed strengthening regulations that require states to identify total maximum daily loads (TMDLs), which estimate how much pollution water bodies can receive without violating water quality standards. These regulations, placed on hold by the administration of President George W. Bush, could impact nonpoint-pollution-generating forestry and agricultural activities not currently subject to federal nonpoint pollution regulations (Copeland 2001).

Nonpoint source pollution results from five major types of land use: agriculture, silviculture, mining, construction, and urban activities, and from atmospheric deposition (Neary *et al.* 1988). Silviculture, urban development, and agriculture are the focus of this project. Between 1988 and 1998, agriculture and urbanization were the leading sources of water quality impairment in the Southeast, while silviculture ranked eighth out of eleven major impairment sources (Table 1.1). Through impacts that include stream channel alternation and changed water and contaminant transport pathways, urbanization increases stream turbidity, nutrient enrichment, bacterial contamination, organic matter loads, concentrations of toxic compounds, and water temperature (Snodgrass *et al.* 1997). In the Southeast, agricultural pollution comes from field tillage, pesticide and fertilizer applications, drainage, grazing, and feed-lot operations. Although little information exists on the cumulative impacts of timber harvesting on overall watershed health (Fulton and West 2002), forestry is considered a less pervasive nonpoint source than agriculture, in part because silvicultural activities occur less frequently and over a smaller area at any given time. Still, forestry practices such as harvesting, site preparation, reforestation, drainage, prescribed burning, road construction, fertilization, and herbicide use all have the potential to pollute

and are considered nonpoint sources (Hohenstein 1987, Sopper 1975). In fact, between 1988 and 1998, an average of 3,600 miles of Southeastern streams was potentially impacted each year by forestry-related pollution (Fulton and West 2002).

Table 1.1: Annual average contribution of point and nonpoint sources of pollution to impaired river miles from 1988 to 1998 in the South. (Source: West 2002).

Source	Miles	Percent	Pollution type
Agriculture	44,326	43.04	nonpoint
Municipal	11,043	10.72	point
Storm sewers/runoff	10,852	10.54	point
Hydrological/habitat modification	8,153	7.92	nonpoint
Industrial	6,990	6.79	point
Resource extraction	6,554	6.36	nonpoint
Land disposal	3,929	3.82	point
Silviculture	3,639	3.53	nonpoint
Construction	3,434	3.33	nonpoint
Channelization	2,337	2.27	nonpoint
Natural	1,723	1.67	nonpoint
Total nonpoint	70,168	68.14	
Total point	32,814	31.86	
Total	102,980	100.00	

Forestry sources of nonpoint pollution are of special concern in areas with valuable and sensitive fisheries (Brown *et al.* 1993). The relationship between forests and aquatic ecosystems is particularly important in the 13 states of the Southeast, where forest covered 214.8 million acres in 1999 (Conner and Hartsell 2002), or about 40 percent of the total land area, or 58 percent when less-forested Texas and Oklahoma are excluded. This forestland is vital for the maintenance and improvement of water quality in the region. Forested watersheds are consistently associated with lower nutrient and sediment inputs, and better aquatic biological conditions, than watersheds that are not forested (West 2002).

The rivers and streams of the Southeast harbor an extraordinary variety of life, resulting from diverse physical geography, a favorable climate, and a dynamic natural history

(Chaplin *et al.* 2000). The Southeast, for example, is home to half the 800 fish species native to North America north of Mexico (Warren *et al.* 1997). In addition, 91 percent of the world's freshwater mussel species and more than half the known fingernail clam and snail species occur in the region (Neves *et al.* 1997). Nonpoint pollution in the Southeast threatens globally significant aquatic biodiversity, including many uncommon and endemic species that inhabit streams and rivers. Of the region's 176 rare aquatic insect species, for example, 82 percent are found in streams and rivers, as are 88 percent of 165 rare fish species, and 57 percent of 159 rare crustaceans (Herrig and Shute 2002). North Carolina alone contains all or part of 22 river basins considered by The Nature Conservancy as critical for the nationwide conservation of at-risk fish and mussel species (Chaplin *et al.* 2000).

The control of nonpoint pollution – which occurs in large open systems and is therefore not traceable to a discreet facility – is more complex than the management of point sources and therefore requires a fundamentally different approach (Ice 1988, Neary *et al.* 1988, Solomon 1988). Riekert *et al.* (1988) noted that the only way to entirely prevent nonpoint pollution from forestry operations is to refrain from any significant activity. However, the use of best management practices (BMPs) is considered an adequate measure for avoiding extensive pollution and promoting the rapid recovery of water quality to pre-activity levels.

BMPs are defined as the methods, measures, or practices determined by a state to be practical and effective in preventing or reducing nonpoint source pollution to a level compatible with water quality goals. BMPs include such structural controls as proper road construction, stream crossings, and the protection of riparian zones from certain activities; correct operation and maintenance procedures; the and scheduling and distribution of

activities (Cubbage *et al.* 1993, DeForest *et al.* 1990, Lynch and Corbett 1990). They generally are designed to prevent negative water quality impacts before they occur, and provide a degree of certainty to forest landowners and managers about their responsibility to protect water quality (Ice 1998).

The BMP considered most effective in the reduction of nonpoint pollution is the use of streamside management zones, also known as riparian buffers, buffer strips or filter strips. Streamside management zones (SMZs) are areas, often with undefined boundaries, that are adjacent to perennial, intermittent, or ephemeral watercourses and are where the biological and physical attributes of the zone ameliorate the impacts of upland activities on the watercourse (Nutter and Gaskin 1988). SMZs protect water quality and habitat for aquatic species by providing shade, producing organic debris, and regulating sediment and nutrient flows from upland areas, through the biological and physical action of the vegetation in the zone as well as chemical, physical and biological action in the soil. They also help stabilize stream banks, provide a source of spawning gravel for fish, moderate riparian microclimate, provide wildlife habitat, and enhance the aesthetic and recreation values of water bodies (O’Laughlin and Belt 1995, Comerford *et al.* 1992).

Other forestry BMPs address pre-harvest planning; stream crossings in harvest zones; logging road construction; use of skid trails, log decks and landings, and portable sawmills; post-harvest site preparation and reforestation; pesticide and herbicide applications; and prescribed burning (N.C. Division of Forest Resources 1989).

1.2 Forestry BMP Adoption and Compliance

By 2001, all 50 states had adopted forestry BMPs to address nonpoint source pollutant concerns, up from 38 states in 1990 (National Association of State Foresters 2001). All 13 Southeastern states have established BMPs to reduce nonpoint pollution from a variety of sources, including forestry. Compliance with silvicultural BMPs is mostly voluntary in all but four Southeastern states – Florida, North Carolina, Virginia, and Kentucky. In the first three, BMP implementation is “quasi-regulatory” because of links to other state regulatory programs, while BMP compliance is mandatory in Kentucky. All Southeastern states have trained landowners, loggers, and forestry practitioners in the proper implementation of these practices (Prud’homme and Greis 2002).

Adherence by forest landowners and timber harvesters to state-adopted forestry BMP rules is reported as high across the country, in the Southeast, and in North Carolina. The compliance rate averages about 86 percent, with twenty-two states reporting greater than 90 percent compliance (National Association of State Foresters 2001). Twelve Southeastern states have measured BMP compliance using unique approaches, reporting broad application during silvicultural operations (Prud’homme and Greis 2002). In general, implementation tends to be lower on private land – with the exception of industrially owned land – than on public forests, and lower on small private tracts of forest than on large private tracts. Implementation rates are generally higher with assistance from a professional forester (Prud’homme and Greis 2002, Ellefson *et al.* 2001).

In North Carolina, the Division of Forest Resources adopted and published forestry BMP manuals for upland sites in 1989, and for wetland sites in 1990. The division is currently rewriting both documents, and may partition the upland manual into sections

specific to North Carolina's mountain, piedmont and coastal plain regions (N.C. Division of Water Quality 2000i).

Before 1990, North Carolina forestry activities were exempt from the state's Sedimentation Control Act (SCA), and BMPs were voluntary. In 1989, state lawmakers voted to maintain the SCA forestry exemption on the condition that all site-disturbing activities adhere to the state's recommended Forest Practices Guidelines. As a result, landowners, timber buyers, and forest operators who are out of compliance must file a sedimentation control plan and meet all its requirements before continuing site-disturbing activities, and may have to pay penalties (Deforest *et al.* 1990). Additionally, the North Carolina Environmental Management Commission requires the maintenance of existing 50-foot forested buffers along perennial and intermittent streams in two river basins deemed to have "nutrient sensitive waters," and part of a third. Those buffer rules went into effect for the Neuse River basin in 1997, and the Tar-Pamlico River basin in 2000 (N.C. Division of Water Quality 2000h). A temporary rule requiring buffers for parts of the Catawba River basin went into effect in 2001 (N.C. Division of Water Quality 2001d).

The N.C. Division of Water Quality (2000i) reported that 1999 compliance with forestry BMPs in North Carolina was 95.5 percent, although the report notes a lack of uniformity in compliance efforts among the N.C. Division of Forest Resources' county, district, and region divisions where the 3,904 forestry sites were inspected.

1.3 Forestry BMP Effectiveness

Monitoring compliance with forestry guidelines and rules is less difficult and less expensive than measuring how effectively those practices help maintain water quality

standards (Ice 1988). While it is important to monitor whether BMPs are being applied as intended, compliance monitoring by itself does not inform regulators and researchers whether BMPs – even with high rates of implementation – result in the desired water quality goals. A focus on compliance monitoring alone also ignores the possibility that water quality standards might be met through the use of practices that are less costly to forest landowners and operators.

While meeting their environmental protection objectives, regulatory policies such as BMPs reduce overall timber production and raise costs for people growing timber, including nonindustrial private forest owners (Granskog *et al.* 2002). BMPs are cost-effective when they neither overconstrain nor underconstrain land management. The economic cost of overconstraining land management results from the waste of resources and the ensuing loss of landowner income. Underconstraining land management results in costs from the effect of poor water quality on aquatic organisms and downstream water users, in addition to the costs of loss of site productivity (Brown *et al.* 1993). Streamside management zones are the largest single BMP cost to landowners, because SMZs usually require retention of a certain portion of the merchantable timber (Aust 1994). As a result, there is particular interest in economic efficiency of riparian buffer regulations and guidelines.

According to policy analysts, the credibility of nonpoint source control programs depends in part on the effectiveness of forestry practices in protecting water quality (Adams *et al.* 1995) and on the cost-effectiveness of those measures in meeting water quality standards (Brown *et al.* 1993). Effectiveness monitoring allows researchers to make informed recommendations to decisionmakers about whether BMPs or water quality standards should be revised, based on (1) whether BMPs are being implemented as designed,

(2) whether practices are meeting the desired water quality objectives, (3) whether the designated beneficial uses of water are being protected, and (4) whether water quality is changing through time (Solomon 1988).

Extensive research indicates that silvicultural BMPs – especially streamside buffers (for example, Comerford *et al.* 1992, Deforest *et al.* 1990, Phillips 1989, Nutter and Gaskin 1988) – are highly effective at reducing sediment-related nonpoint pollution and at preventing drastic changes in stream water temperature. Buffers and other management practices are also believed to lower nutrient, pesticide and herbicide flows into streams and lakes, although the science remains sketchy in this area. In the Southeast, properly implemented and maintained BMPs are considered highly effective in controlling a variety of nonpoint pollutants, especially in areas with steep topography (Fulton and West 2002).

Complicating the picture is the lack of BMP consistency among Southeastern states. Aust (1994), for example, found no consensus on the details of BMPs – especially for streamside management zones – among forestry BMP manuals published in eight Southeastern states. At least 11 of the 13 Southeastern states encourage or require the protection of SMZs as a best management practice during silvicultural activities, although the width of these stream buffers varies by state (Prud'homme and Greis 2002). This inconsistency may result, at least in part, from insufficient research on the proper riparian buffer widths to prevent the excessive flow of nutrients and other chemical solutes into water bodies (Comerford *et al.* 1992, Nutter and Gaskin 1988).

What researchers do know about sufficient stream buffer widths is that they vary considerably based on site conditions, and on the water quality or aquatic ecology attribute under consideration. Investigating run-off detention time in Piedmont North Carolina stream

buffers, Xiang (1996) found that the desirable buffer width varied significantly depending on slope, soil features, and land surface conditions. This width ranged from a minimum of 7.9 meters to a maximum of 176 meters. A literature review by Large and Petts (1994) found that the recommended width for sediment control varied from 15 to 213 meters, and from 1 meter to 150 meters for general water quality control. For fisheries protection, widths spanned 10 to more than 20 meters. Despite this variability in buffer width, regulators and land managers typically set fixed buffer widths to simplify implementation and reduce costs.

Clearly, more study is needed to determine whether BMPs – principally riparian buffers – result in water quality conditions that meet required standards, especially when coupled with upland land-use changes. Such analysis is especially useful at the landscape scale, the scale at which cumulative effects become apparent (Graham *et al.* 1991); these cumulative effects are defined in National Environmental Policy Act (NEPA) regulations as “the impact on the environment which results from the incremental impact of the action when added to other past, present, and reasonably foreseeable future actions, regardless of what agency (Federal or non-federal) or person undertakes such other actions” (40 CFR § 1508.7). Evidence is increasing that the most devastating environmental damage may result from the cumulative effects over time of multiple minor actions, rather than from the direct effects of a single major action (U.S. Council on Environmental Quality 1997).

Additionally, policymakers and land managers usually make regulatory and management decisions at the scale of counties, river basins, and states. Ecological risk assessment appears to lend itself well as a tool for complicated research issues at these scales.

1.4 Ecological Risk Assessment Overview

Researchers have noted the difficulty of quantifying and establishing cause-and-effect relationships between nonpoint pollution sources and water quality degradation. This is because 1) nonpoint pollution enters water bodies intermittently from a large area, often during and following precipitation events; 2) it moves through the landscape in a diffuse manner, both above and below ground; 3) nonpoint pollution caused by human activities is difficult to distinguish from natural background loads of certain pollutants; and 4) it can be discharged into water bodies long after its initial entry into the landscape (Neary *et al.* 1988, Ice 1998). These factors are more troublesome when examining relationships at a landscape scale – such as a watershed or basin – than at the site scale, in part because the characteristics of streams are influenced by a variety of correlated landscape features (Richards *et al.* 1996). At the same time, it is often better to focus on the regional or landscape scale rather than the local scale in studying such problems, because many cumulative effects of human disturbance are especially apparent at the regional scale (Graham *et al.* 1991).

To understand the relationship between human alteration of the landscape and nonpoint pollution, it is important to separate natural landscape characteristics from human-related disturbance over which policymakers and land use managers may be able to exercise regulatory control. Specifically, the range of human activities and the risks they pose to aquatic ecosystems should be identified by assessing relative risk and by placing sample sites on a human disturbance gradient (Byrce *et al.* 1999). It is important, however, to recognize two important limitations: The cost of eliminating all the environmental effects of human disturbance is excessively high, and regulatory decisions have to be made using incomplete scientific information (Ruckelshaus 1983).

Ecological risk assessment is an approach that holds promise for providing useful answers about how human decisions impact complicated aquatic ecosystem interactions (Zandbergen 1998, Chen *et al.* 1993). Risk assessment is the process of assigning magnitudes and probabilities to the adverse effects of human activities or natural catastrophes (Suter 1993). It is a methodology employed for more than a century in economic fields such as banking and insurance and, only slightly more recently, in human health and safety studies and disaster management. The contemporary, quantitative application of risk assessment relating to the regulation of environmental pollutants began in the mid-1970s. With the growing realization that watershed and ecosystem degradation is caused by physical, biological, and chemical factors, a separate field has developed focusing on more complex ecological risks to ecosystems (Renner 1996). This field of study stems from the understanding that healthy ecosystems provide a wide array of products and services to humans, including renewable resources and food, water storage and flood control, biodegradation and removal of contamination from air and water, pest and disease control, moderation of climatic extremes, recreational opportunities, and scenic beauty (Borouh 1998).

Although the use of ecological risk assessment has been mainly to estimate the risks posed by chemicals introduced into the environment (Borouh 1998), it also has promise for efforts to control water pollution. For example, Chen *et al.* (1993) note that the 1977 Clean Water Act explicitly mandates the maintenance and restoration of biological integrity – a management goal they contend has been long neglected. They argue that since ecological integrity is the paramount goal of water quality protection, ecological risk assessment – using water quality modeling methods – should be the new paradigm for achieving this long-term

goal. Another researcher studying the impact of urbanization in watersheds (Zandbergen 1998) called ecological risk assessment an integrative approach that balances the complexity of scientific analysis with land managers' need for clear and simple answers about the condition of the watershed and the actions needed to achieve certain objectives.

Chapter 2

Ecological Risk Assessment Methodology

Ecological risk assessment is an integrated method for analyzing and predicting ecosystem response to stress, which occurs in the form of such stressors as pollution or habitat destruction. It is a process used to systematically evaluate and organize data, information, assumptions, and uncertainties to help understand and predict the relationships between stressors and ecological effects in a fashion useful for environmental decision-making (U.S. Environmental Protection Agency 1998a). Formal quantitative techniques are used to estimate the probabilities of certain effects on well-defined endpoints and to estimate uncertainties associated with the analysis (Suter 1993). Endpoints are defined as explicit expressions of the environmental value to be protected, operationally defined by an ecological entity and its attributes (U.S. Environmental Protection Agency 1998a). This process is clearly separated from that of choosing among alternatives and determining the acceptability of risks, which is the responsibility of policymakers (Suter 1993).

Human health risk assessments focus on the risks of chemicals to individuals within exposed human populations. In contrast, ecological risks result from exposure of populations, communities, and ecosystems to stresses acting individually or in combination at diverse spatial scales (Gentile and Slimak 1992). The central objective of ecological risk assessment is to estimate the possibility of adverse impacts on ecosystems from exposure to stress sources such as technology (including roads and the development of land) and pollutants (Borouh 1998). It should provide a quantitative basis for balancing and comparing risks associated with environmental problems, and a systematic means of improving the estimation and understanding of those risks (Graham *et al.* 1991). The scope of such an assessment can range from site-specific to international, as in the case of worldwide climate change. In its

methods, ecological risk assessment draws widely on the standard procedures of environmental impact analysis and monitoring (Borouh 1998).

The U.S. Environmental Protection Agency's official guidelines on ecological risk assessment (1998a) divide the process into four stages (Figure 2.1). The first two stages – problem formulation and risk analysis – clearly define the sources of the problem, the endpoints of the assessment, and the mechanisms by which the hazard may affect the endpoints. This process may result in the discovery of gaps in knowledge about the problem, and can therefore identify further research needs (Graham *et al.* 1991). In the third stage – risk characterization – the risk assessor estimates the spatial and temporal patterns of exposure to the hazard, and quantifies the relationship between exposure and effects to determine risk (Graham *et al.* 1991). The fourth step is the communication of assessment results.

The problem formulation stage provides a foundation for the rest of the assessment process by generating and evaluating preliminary hypotheses about why ecological effects have occurred, or may occur, as a result of human activities. The three products of this phase are 1) assessment endpoints that adequately reflect management goals and the ecosystem they represent, 2) conceptual models that describe key relationships between a stressor and an assessment endpoint, or between several stressors and endpoints, and 3) an analysis plan.

The assessment endpoints and conceptual models provide the focus and structure for the analysis phase. This flexible second stage in the assessment process produces summary profiles that describe the relationship between the stressor and response. Also in this phase, the risk assessor selects data based on their usefulness for evaluating the risk hypotheses.

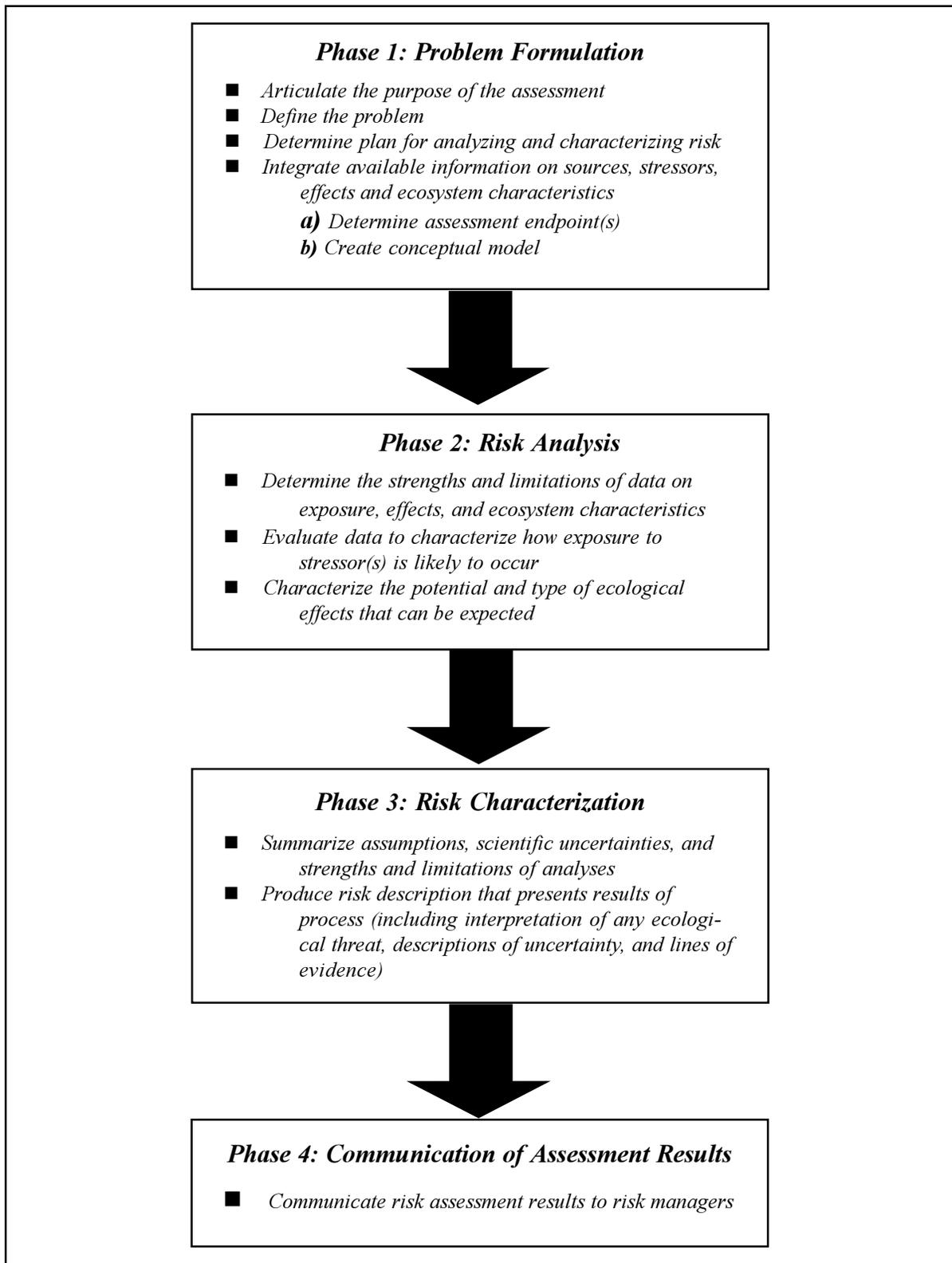


Figure 2.1: The Risk Assessment Framework (modified from U.S. EPA 1998a)

Risk characterization is the culmination of this process, allowing the risk assessor to clarify the interactions among stressors, effects, and ecological entities under study, and to reach conclusions about exposure to risk and the degree of existing or future effects. This means using the results of the analysis phase to develop an estimate of the risk posed to the assessment endpoints identified in the problem formulation phase. The assessor also identifies and summarizes the uncertainties and assumptions of the risk assessment, which are then reported to risk managers and others in the communication phase of the process (U.S. Environmental Protection Agency 1998a). Managers and elected officials can then integrate results of the risk assessment with economics, social values, regulatory strategies, and policy in making a decision (Gentile and Slimak 1992).

The ecological risk assessment process can be an iterative one, with new information and results from later steps being used to revisit earlier steps or to reinitiate the process. Such iteration can result in improved environmental decision-making, and is consistent with adaptive management principles often used in managing natural resources (U.S. Environmental Protection Agency 1998a).

To fulfill the need for better management decisions, risk assessments need to offer a quantitative basis for balancing and comparing risks associated with environmental hazards, and a systematic means of improving the estimation and understanding of those risks. Uncertainties concerning potential environmental effects are explicitly recognized and, if possible, quantified (Hunsaker *et al.* 1990).

Retrospective ecological risk assessments seek to establish the relationship between a pollution source – which may be unknown or have uncertain effects, as is the case with nonpoint source pollutants – and an ecological effect (Hession *et al.* 1996, Suter 1993).

Prospective assessments, meanwhile, aim to predict the impact of potential stresses that have not yet occurred (Gentile and Slimak 1992).

An integrated, top-down, and often retrospective approach to ecological risk assessment is particularly well suited to evaluating the multiple and cumulative effects of chemical and non-chemical stresses at regional scales (Gentile and Slimak 1992), the scale at which cumulative effects are often more apparent (Graham *et al.* 1991).

2.1 Ecological Risk Assessment Strengths and Weaknesses

Ecological risk assessment is still a relatively new methodology, and is somewhat controversial because of its newness and complexity (Renner 1996). As recently as the mid-1990s, Chen *et al.* (1994) called ecological risk assessment “almost a virgin area” in determining how to control nonpoint source pollution generated by land-use activities. Proponents characterize it as a useful tool allowing policymakers and managers to compare and prioritize the risk of various decisions. Critics contend that the methodology is overly laden with value decisions, and only proves useful with highly constrained problems that may not be of use to decision-makers.

The EPA’s ecological risk assessment guidelines (1998a) list the following advantages of the methodology:

- The results of an assessment can provide a basis for quantitatively comparing, ranking, and prioritizing risks. This can be useful in cost-benefit and cost-effectiveness analyses of alternative management options.

- Through an iterative process, new information can be incorporated into risk assessments, leading to better environmental decision-making.
- Risk assessments can express changes in ecological effects as a function of changes in exposure to stressors. This could be useful to decision makers who are required to evaluate tradeoffs, examine different alternatives, or determine the extent to which stressors have to be reduced to achieve a given outcome.
- Ecological risk assessments are unique in providing a scientific evaluation of ecological risk that explicitly addresses uncertainty. Uncertainty analysis allows the assessor to describe the degree of confidence in an assessment, and can help policymakers focus research on areas that will lead to the greatest reductions in uncertainty.

Suter (1993) outlines three additional ecological risk assessment strengths:

- By expressing results as probabilities, risk assessment acknowledges the inherent uncertainty in predicting future environmental states, making an assessment more credible.
- Ecological risk assessment provides a systematic means to improve the understanding of risks, based on uncertainties in the process.

- By basing the estimates of probability and the magnitudes of effects on formal quantitative methods, risk assessment provides a tool for the parties involved in environmental decisions to compare the implications of their assumptions and data rather than negotiating on the basis of clout.

The first – and perhaps most difficult – challenge risk assessors face in the ecological risk assessment process is defining and characterizing the focus of the assessment, the environmental attribute that is subjected to stress. This requires the identification and selection of appropriate ecological endpoints and indicators for measuring the effects of the stress and assessing its consequences (Gentile and Slimak 1992). Selecting the impact on which to focus the assessment is at least partially a decision tied to individual values of what ecological changes are adverse, and therefore undesirable, or beneficial and desirable (Borouh 1998, Lackey 1997a, Lackey 1997b). Different people will have different views about what constitutes an unacceptable impact, which illustrates that ecological risk assessments are social as well as scientific activities (Renner 1996). The term ecosystem “health” is particularly troublesome to some skeptics because, they explain, it is inadequately quantified and overly value-laden (Borouh 1998, Lackey 1997a).

In many cases, however, the values of risk assessors are in keeping with the values of society, pluralistic and divided as it may be. Americans have spoken on many of these issues – through their elected representatives – with three decades of laws that declare clean water and the protection of endangered species, among other things, as having positive value. As Lackey (1997b) notes, risk assessment is useful in ecological management situations where a legislative or policy basis exists defining what is ecologically adverse.

Additionally, the ecological risk assessment process clearly separates risk analysis – the scientific process of estimating the size and probability of effects – from risk management – the process of choosing among alternatives and determining the acceptability of risks (Suter 1993). As a result, the risk assessment is less likely to be biased by the values of the risk assessor. And, as Suter (1993) points out, ecological risk assessments should estimate the impact on clear and consistent endpoints rather than such ambiguous endpoints as “ecosystem health.”

Another criticism is that performing credible risk assessments for complex ecological problems is difficult unless the boundaries of the assessment problem are highly constrained. However, narrowly defining ecological problems produces risk assessments that are of limited relevance in resolving public policy questions (Lackey 1996). In other words, anything that can be easily and quickly measured with precision is probably irrelevant in ecosystem management (Lackey 1997b). Even defined in narrow terms, the assessment may be technically complex and require sophisticated scientific information and analysis (Lackey 1997a).

At the same time, risk assessments often consider management goals and objectives as well as scientific issues in the development of assessment endpoints and conceptual models. According to the EPA’s ecological risk assessment guidelines, this initial planning helps ensure that the results will be useful to risk managers (U.S. Environmental Protection Agency 1998a).

Clearly, completing an ecological risk assessment is a balancing act that requires capturing the essential features of a hazard within the constraints of data availability. Because region-specific and geographically explicit data are often not available, for example,

a regional ecological risk assessment may fall short of its goal. Nevertheless, the process of an assessment should lead to improved estimation and understanding of regional risks and, hopefully, better policy decisions (Graham *et al.* 1991).

If nothing else, risk assessment can be valuable in decision-making if it forces debate about the initial policy question or problem shaping the assessment (Lackey 1997a), and about the costs and benefits, winners and losers, and the ecological consequences of the range of decision options (Lackey 1997b).

2.2 Project Overview

The central goal of this project is the development of a set of detailed vulnerability models to help policymakers and natural resource managers understand the impact of land cover changes on water quality in North Carolina. Specifically, I compared which landscape characteristics at the scale of the watershed, or of riparian zones in that watershed, are the best predictors of stream invertebrate community integrity. This, in turn, can be used to document spatial and temporal changes in water quality (N.C. Division of Water Quality 2001a). The results will allow managers and policymakers to weigh the risks of management and policy decisions to a given watershed or set of watersheds, including whether streamside buffer protection zones are ecologically effective and economically efficient in achieving water quality standards.

In this study, I utilized an approach similar to the one employed by Diamond and Serveiss (2001), who used a stepwise multiple regression analysis of GIS data to identify the sources of stress to fish and rare and endangered mollusks in the Clinch and Powell River Basin of southwestern Virginia. I selected the landscape variables analyzed in the

assessment after following the steps recommended by the U.S. Environmental Protection Agency for an ecological risk assessment (U.S. Environmental Protection Agency 1998a). I integrated the available information on sources, stressors, effects, and ecosystem characteristics related to the assessment endpoint (macroinvertebrate community structure), and determined the strengths and limitation of the data. I used stepwise regression to select the best landscape predictors of benthic macroinvertebrate water quality tolerance scores. Point sources of pollution, which account for about 30 percent of water quality degradation in the Southeast and 50 percent nationwide (West 2002), were not included in this analysis because of a lack of adequate data on the sources of such pollution across North Carolina.

To facilitate extrapolation among watersheds within a region and across regions, ecological risk assessments should be based on a model describing the underlying factors that influence watershed response to sources of stress, particularly vulnerability to change as a result of those stresses (Detenback *et al.* 2000). After assembling such a model for this study, I measured 11 land cover and land form characteristics for 74 North Carolina watersheds, several at both the watershed scale and riparian scale (zones 300, 100, and 50 feet on either side of streams). The regression analysis considered which of these landscape characteristics have a significant impact on the integrity of stream life, as measured by two metrics describing the tolerance of benthic macroinvertebrates to stream degradation. These indices are the North Carolina Biotic Index (Lenat 1993), and an index of Ephemeroptera (mayfly), Plecoptera (stonefly), and Trichoptera (caddisfly) tolerance (N.C. Division of Water Quality 2001a).

The result of the regression analysis is a vulnerability equation allowing managers and others to predict the condition of stream invertebrate metrics – and, therefore, water

quality – at any freshwater stream sampling site in North Carolina, by measuring only a few landscape characteristics associated with the site’s catchment. By inputting variable data in the model equation to reflect proposed landscape changes (such as the harvest of a certain percent of forest in the watershed, or the use of a certain riparian buffer width), the model user can better understand potential water quality changes. The output of the equation would be a score for the site and its catchment, with 0 indicating the presence of stream invertebrates least tolerant of degradation (i.e., the highest water quality), and 10 indicating an invertebrate community most tolerant of degradation (i.e., the lowest water quality).

Ideally, this approach could be used throughout the Southeast, with different equations derived for states or ecoregions such as the Southern Appalachians or Piedmont.

2.3 Ecological Risk Assessment Methodology

As outlined in the U.S. Environmental Protection Agency’s official guidelines (1998a), ecological risk assessment is a four-phase process (Figure 2.1) that encompasses problem formulation, risk analysis, risk characterization, and communication of assessment results. I used this general framework to analyze the risk of landscape-scale land-use change in North Carolina to the state’s aquatic biota.

2.3.1 Phase 1: Problem Formulation

The first phase of the ecological risk assessment framework has four major components: (a) articulating the purpose of the assessment, (b) defining the problem, (c) determining a plan for analyzing and characterizing risk, and (d) integrating available information on sources, stressors, effects and ecosystem characteristics.

a) Assessment Purpose

Compliance with voluntary and semi-voluntary forestry best management practices (BMPs) is considered high throughout the United States (National Association of State Foresters 2001). It is unclear, however, whether BMPs effectively protect water quality from nonpoint source pollution in the Southeast. This assessment, therefore, aims to provide important information about the relationship in North Carolina between landscape characteristics – especially land use – and water quality. The emphasis is on Streamside Management Zones (SMZs), also known as riparian buffers, because they represent the principal best management practice that can be measured and analyzed at the landscape scale.

b) Problem Definition

Nonpoint pollution may be responsible for as much as 50 percent of water quality degradation nationally (Copeland 2001), for 70 percent in the Southeast (West 2002), and for 64 percent in North Carolina (N.C. Division of Water Quality 2000a). Urban development, agriculture, and silvicultural activities all contribute to the reduction of water quality and degradation of aquatic life through the runoff of sediment, nutrients, and other nonpoint pollutants. This is a particular concern in the Southeast, a region with extensive forestlands, quickly expanding populations, and high-quality aquatic resources.

c) Risk Analysis and Characterization Plan

The risk analysis and characterization plan for this project was completed February 25, 2002, and presented to my graduate committee for review and discussion.

d) Integration of Information on Sources, Stressors, Effects, and Ecosystem Characteristics

The U.S. Environmental Protection Agency (1998a) divides this part of problem formulation into two stages: determining an endpoint, and creating a conceptual model.

1) *Endpoint Determination*

Ecological risk assessment endpoints are formal expressions of the actual environmental value to be protected (Suter 1990). Choosing a proper endpoint may be one of the most difficult steps in the assessment process (Renner 1996).

Ecological risk assessments must have clearly defined endpoints that are biologically and socially relevant, accessible to prediction and measurement, and susceptible to the hazard being assessed (Suter 1990). In this assessment, water quality and aquatic ecological integrity are the values to be protected; the community structure of benthic macroinvertebrates, which are associated with the substrates of aquatic ecosystems, was chosen as the endpoint for this assessment because it meets most, if not all, of the criteria listed above. Additionally, the species composition of an aquatic community has been suggested as an appropriate endpoint for a regional aquatic assessment, which requires endpoint observations often over large geographic areas and often long time periods (Hunsaker *et al.* 1990).

Macrobenthic aquatic invertebrates – which include the aquatic nymphs and larvae of such insect orders as Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies) – are considered especially good indicators of both short-term and long-term changes in water quality. Because macrobenthic invertebrates are less mobile than other

types of aquatic organisms, their community structure responds to a wide variety of pollutants. Because many species have life cycles longer than a year, these communities can reflect both short-term and long-term trends (N.C. Department of Water Quality 2001a). Additionally, these stream invertebrates are indicators of nonpoint source pollution impacts from such land-use activities as silviculture because they are abundant in low-order streams where much timber harvesting occurs, and are sensitive to habitat and water quality changes (Adams *et al.* 1995). They are also biologically crucial in the food web of aquatic ecosystems (Giller and Malmqvist 1998), although their social relevance to humans is arguable.

Since 1983, the North Carolina Division of Water Quality (NCDWQ), a section of the state Department of Environment and Natural Resources, has collected more than 5,000 macrobenthic invertebrate samples from roughly 2,100 stream and river sites throughout North Carolina. These sites were chosen to monitor all of the state's larger rivers and streams, to test water quality above and below wastewater treatment plants, and to detect changes in water quality following land use changes (David Lenat, pers. com.). Benthic macroinvertebrate data are gathered during the compilation of the water quality assessment reports that the NCDWQ assembles every five years for each of the state's 16 major river basins. The resulting information, which complements water chemistry analysis by NCDWQ scientists, is used to document temporal and spatial changes in water quality (N.C. Division of Water Quality 2001a).

Personnel from the NCDWQ's Biological Assessment Unit use a standard qualitative sampling procedure to collect macrobenthic invertebrates from wadeable freshwater streams and rivers. This procedure comprises ten composite samples: two kick-net samples, three

bank sweeps, two rock or log washes, one sand sample, one leaf pack sample, and visual collections from rocks and logs. Using forceps and white plastic trays, the field workers remove a subset of invertebrate organisms from the sample in rough proportion to their overall abundance; these are then placed in glass vials containing 95 percent ethanol and are returned to the Biological Assessment Unit in Raleigh. Sampling typically requires four to six hours for one collector, or one-and-a-half to two hours for three collectors (N.C. Division of Water Quality 2001a, 2002a).

After the organisms are identified to the lowest possible taxonomic level, Biological Assessment Unit personnel calculate a variety of data summaries to describe the sample, including the total invertebrate taxa richness; the richness and abundance of EPT (Ephemeroptera, Plecoptera, and Trichoptera) taxa; and two biotic index values. These metrics are then used to assign a water quality classification – Excellent, Good, Good/Fair, Fair, or Poor – to the sample site (N.C. Division of Water Quality 2001a).

For this study, two metrics of invertebrate water quality tolerance were chosen as endpoints: the North Carolina Biotic Index, and the EPT Biotic Index.

The North Carolina Biotic Index (NCBI) was developed by Lenat (1993) to examine the general level of pollution at stream sites; his approach was modeled after the biotic index Hilsenhoff (1987) created for use in Wisconsin streams and rivers. The NCBI rates stream sites based on the water quality tolerance of the macroinvertebrates sampled at the site; Lenat determined each taxon's level of pollution tolerance by studying more than 2,000 macroinvertebrate stream samples collected in North Carolina between 1983 and 1992. Taxa found in higher numbers at degraded sites were given higher tolerance values, since they

tolerate higher levels of pollutants; taxa found in greater numbers at less degraded sites were assigned lower tolerance values.

The NCBI score for a site measures the degradation tolerance of the benthic macroinvertebrates, relative to their abundance, and is derived using the following equation (Lenat 1993):

$$BI = \frac{\sum (TV_i \cdot n_i)}{N}$$

where:

BI is the biotic index value for the site.

TV_i is the *i*th taxon's tolerance value (from 1 to 10), based on the average abundance of each taxon in each of five water-quality categories. The abundance values were converted into cumulative percentiles and graphed versus the water-quality score categories (1=excellent, 2=good, 3=good-fair, 4=fair, and 5=poor). The 75th percentile produced the greatest separation between tolerant and intolerant species, while leaving facultative species with values near the mid-range. Therefore, the water-quality score at the 75th percentile (for example, 4.04 is between fair and poor) was interpolated for each taxon. This value was then converted to a scale of 1 to 10, using the equation

$$Final\ TV = 2(1.43 \times Preliminary\ TV - 1.43).$$

n_i is the i th taxa's semiquantitative abundance value in the sample (1 if the taxon is rare [1-2 per sample], 3 if it is common [3-9 per sample], or 10 if it is abundant [10 or more per sample]).

N is the sum of all abundance values in a sample.

The result is a score on scale of 1 to 10, with smaller numbers indicating lower invertebrate tolerance to pollution and, therefore, better water quality. Larger numbers are associated with more degraded stream conditions.

EPT Biotic Index scores (N.C. Division of Water Quality 2001a) are calculated the same way as NCBI scores, but are restricted to three invertebrate taxa considered highly sensitive to water quality degradation: the aquatic nymphs of Ephemeroptera (mayflies) and Plecoptera (stoneflies), and the larvae of Trichoptera (caddisflies).

2) Creation of Conceptual Model

The stated objective of the 1977 amendments to the 1972 Federal Water Pollution Control Act – more widely known as the Clean Water Act – is the “restoration and maintenance of chemical, physical and biological integrity of the Nation’s waters” (33 U.S.C. § 1251). The chemical and physical integrity of water is widely understood as the lack of excessive negative chemical, sediment, and temperature changes; biological integrity has been defined by the U.S. EPA (1990) as the condition of the aquatic community inhabiting the unimpaired water bodies of specific habitat as measured by community structure and function. When existing simultaneously, all three of these characteristics – including aquatic

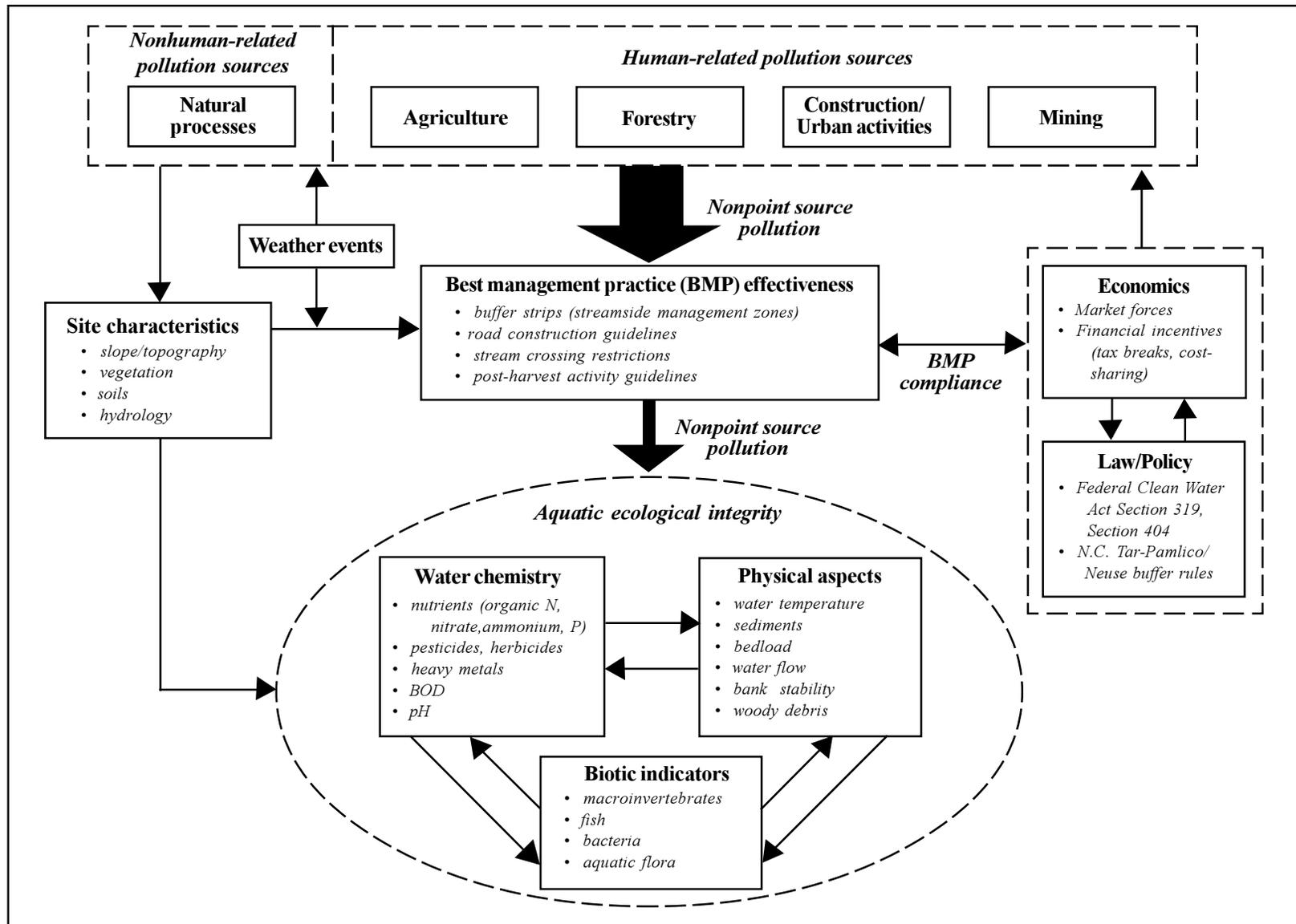


Figure 2.2: The interrelationship of nonpoint source pollution, best management practices (BMPs), and aquatic ecological integrity

habitat –equate to ecological integrity for an aquatic ecosystem; this is considered to be the ultimate goal of water quality protection (U.S. Environmental Protection Agency 1990). Similarly, Karr (1999) defines aquatic ecological integrity, or aquatic health, as the condition of places that support a biota that is a product of evolutionary and biogeographic processes with minimal influence from modern human society. This definition is built around three central principles: (1) a biota spans a variety of spatial and temporal scales; (2) a living system includes an array of kinds of things (the elements of biodiversity) plus the processes that generate and maintain them; and (3) living systems are embedded in dynamic evolutionary and biogeographic contexts (Karr 1999).

Figure 2.2 displays the relationships among nonpoint source pollution, landscape characteristics, human activities, best management practices, and aquatic ecological integrity. Ecological integrity encompasses three components: water chemistry, physical and habitat characteristics, and aquatic life. The community structure of benthic macroinvertebrates, the endpoint of this assessment, reflects the overall ecological integrity of a water body because these organisms respond to episodic and cumulative pollutants and to physical habitat alteration, thus integrating stressors through time (Plafkin *et al.* 1989).

At the same time, site characteristics at both the watershed and riparian scale – such as topography, vegetation, soils, and hydrology – determine aquatic habitat conditions, which shape the composition of aquatic biological communities in concert with the evolved attributes of organisms (Horner *et al.* 1997). Those same site attributes also influence natural processes that release nonpoint pollution into streams.

Nonpoint pollution stresses to ecological integrity have both natural and human origins, although it is often difficult to separate the effects of human land use from a variety

of correlated natural features (Richards *et al.* 1996). Best management practices (BMPs) such as streamside management zones, however, can lessen the impacts of nonpoint pollution on the components of aquatic ecological integrity (Comerford *et al.* 1992, Lynch and Corbett 1990, Nutter and Gaskin 1988, Cooper *et al.* 1987). The use and effectiveness of BMPs, meanwhile, depend on the specific requirements of water quality law and regulation at the federal, state, and local levels, and on economic pressures either encouraging or discouraging BMP use.

2.3.2 Phase 2: Risk Analysis

The second phase of the ecological risk assessment framework – risk analysis – has three steps: a) determining the strengths and limitations of data on exposure, effects, and ecosystem characteristics, b) evaluating data to characterize how exposure to stressor(s) is likely to occur, and c) characterizing the potential and type of ecological effects that can be expected.

a) Determination of the strengths and limitation of data on exposure, effects, and ecosystem characteristics

1) Exposure

Because of the diffuse, often landscape-wide origin of nonpoint source pollution, it is impossible to find any direct data at a statewide scale about the exposure of the endpoints in this assessment – benthic macroinvertebrate community structure – to specific nonpoint pollutants.

2) *Effects*

The North Carolina Biotic Index, one of the endpoints, is considered a more reliable indicator of stream chemistry and habitat quality than of in-stream sediment (N.C. Division of Water Quality 2001a), which is among the more important pollutant sources in the Southeast. The EPT Biotic Index is believed to better measure the impact of sediment in streams (David Lenat, pers. com.). The EPT Biotic Index, however, is considered a less reliable measure of water quality, especially at high elevation sites and at sites where few EPT organisms are collected. Unlike the NCBI, it is not directly used in assigning water quality classifications to sampling sites, but is used to help interpret other data. Scores for both indices vary by ecoregion in North Carolina, with excellent water quality associated with index scores less than 4.05 in the Southern Appalachians, less than 5.19 in the Piedmont, and less than 5.47 in the Coastal Plain sediment (N.C. Division of Water Quality 2001a).

Indices such as these are sometimes criticized for “losing” information about actual ecological conditions. At the same time, they represent a powerful tool for presenting complex information to the public (Messer 1992), including policy and management decision-makers. Considered together, these specific metrics appear to represent the best existing indicators of overall stream water quality in North Carolina. Once other, more-inclusive measures of water quality reduction and stream degradation are developed, they could be integrated easily into this risk assessment methodology.

3) Ecosystem Characteristics

I conducted a literature search to determine the connections between land-use change, other landscape characteristics, nonpoint pollution, and changes in aquatic ecological integrity. It quickly became apparent that the most useful data for investigating these connections were those compiled in a Geographic Information System (GIS) format, as many researchers have done in recent analyses of nonpoint pollution (Diamond and Serveiss 2001, Johnson et al. 2001, Sliva and Williams 2001, Sponseller et al. 2001, Wang et al. 2001, Detenback et al. 2000, Basnyat et al. 1999, Harding et al. 1998, Herlihy et al. 1998, Lent et al. 1998, Zandbergen 1998, Wang et al. 1997, Poiani et al. 1996, Richards et al. 1996, Roth et al. 1996, Weller et al. 1996, Dodd et al. 1994, Kasi 1994, Richards and Host 1994, Hunsaker et al. 1992, Tim et al. 1992).

A wide variety of spatial data are available on landscape characteristics for North Carolina. These data, however, are often inconsistent in scale, geographic projection, and – in the case of land-use data – definitions of land cover classes. Three sets of land-use data from three different years, for example, were considered for this analysis, with a goal of determining the extent of land cover change over time, which could then be compared to changes in macroinvertebrate community structure. The first of these, from the late 1970s and early 1980s, is in vector format, and was developed to be used at the scale of a U.S. Geological Survey quadrangle or larger (U.S. Environmental Protection Agency 1998f). The other two land cover data sets, raster data from 1992 (Multi-Resolution Land Characterization Consortium 2000) and 1996 (N.C. Center for Geographic Information and Analysis 1997), had similar resolutions at about 30 m² per cell, but have incompatible land-use classifications. As a result, the assessment could only be conducted for one point in time:

1992. This dataset was chosen because another, presumably similar, Multi-Resolution Land Characterization (MRLC) dataset is scheduled to be released during the next couple of years.

Some of the individual data sets have shortcomings as well. Spatial stream reach data (U.S. Environmental Protection Agency 1998d) are based on U.S. Geological Survey maps that do not completely reflect the presence of intermittent and perennial streams.

Additionally, they do not include any ephemeral streams, which can contribute significant amounts of sediment and other pollutants to the stream network following storm events (Grey and Henry 2002, Bolton and Ward 1993, Neary *et al.* 1993, Ursic 1991). GIS data on soil characteristics (Schwarz and Alexander 1995) are at a large scale, and lack values for many soil attributes. Precipitation data are also less than perfect because rainfall monitoring stations did not exist near most of the invertebrate sampling sites. Despite these shortcomings, spatial data appear to be the only type useful for quantifying landscape characteristics at the scales at which this assessment was conducted.

b) Evaluation of data to characterize how exposure to stressor(s) is likely to occur

Human-induced modification of aquatic biological systems comes in the form of (1) altered physical habitat, (2) modified seasonal water flow, (3) altered systemic food base, (4) chemical contamination of the water, and (5) changed interactions among organisms (Karr 1999). The first four types of aquatic ecosystem modification result, at least in part, from nonpoint source pollution (Chen *et al.* 1994), while the fifth category, altered organism interactions, may be indirectly related to nonpoint pollution by way of the other changes. Land management, landscape characteristics, and pollutant stressors all affect the assessment endpoint – stream ecological integrity as measured by the two indices of benthic

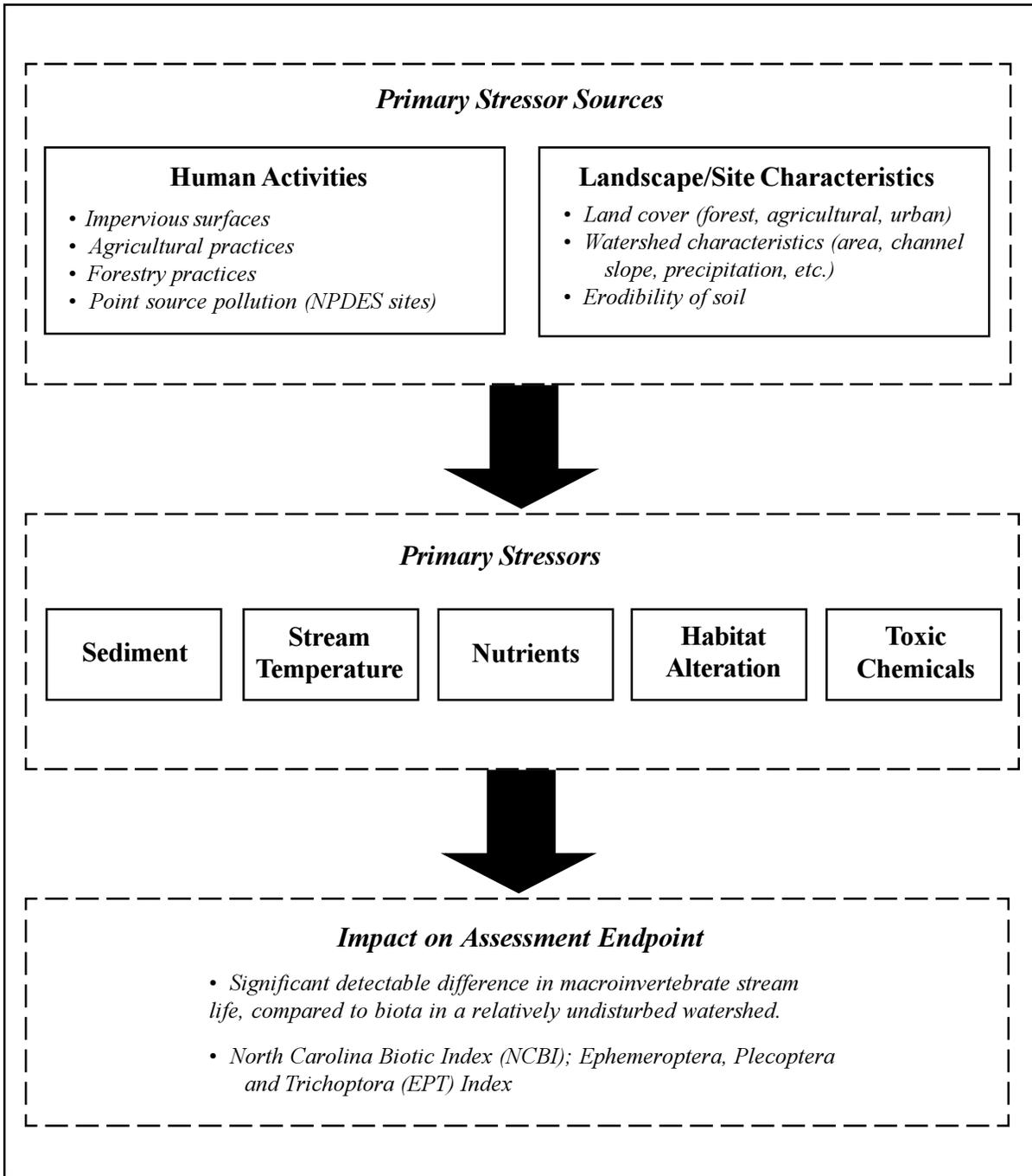


Figure 2.3: The relationships among landscape characteristics and sources of stress on the endpoint of the ecological risk assessment

macroinvertebrate tolerance (Figure 2.3). The primary stressors impacting macroinvertebrate communities are (1) the deposition of sediment into streams, (2) changes in stream temperature, (3) the input of excessive nutrients into aquatic ecosystems, (4) the alteration of stream habitat, and (5) the flow of toxic chemicals in waters inhabited by stream invertebrates. All five of these stressors are either nonpoint pollutants, or are caused at least in part as a result of nonpoint pollution. These stressors come from, and are influenced by, two major sources: human activities, in the form of impervious surfaces, agricultural practices, silvicultural operations, and point source pollution; and landscape and site characteristics, including land cover, watershed characteristics, and soil erodibility.

From a policy perspective, these stressor sources are of considerable interest, because they typically can be measured at a landscape scale, unlike the stressors themselves. In the case of human activities, they often also can be regulated. An extensive amount of data is available on some sources, such as point-source pollution and land use changes. Many of these data are available in GIS format, improving the ease with which they can be analyzed.

Several landscape characteristics are connected, either directly or indirectly, to nonpoint pollution stress on aquatic ecosystems (Table 2.1). Each of these characteristics was included as a variable in the analysis of how these attributes, at the watershed or riparian scale, affect macroinvertebrate index scores. The landscape characteristics come in two forms:

- (1) Land use: the amount of forested, agricultural, and developed land. Land-use attributes are measures of human-related disturbance over which policymakers and land managers may have some influence;

- (2) Land form factors: the types of natural variation that affect the type and effect of impacts of land-use activities.

After transforming several of the variables to achieve a more normal distribution (SAS Institute Inc. 2000), I ran simple linear regressions for each of the landscape variables against the invertebrate index variables, to determine the strength and direction of the relationships. A larger coefficient of determination (r^2) value indicates a stronger relationship between variability in the landscape characteristic and the invertebrate index, while a positive regression coefficient suggests that a landscape variable is associated with invertebrate taxa tolerant of greater stream degradation. A positive regression coefficient, therefore, indicates the landscape variable negatively impacts water quality, while a negative coefficient has a positive effect. (For specific results, see Chapter 3: “A Watershed-Scale Model for Predicting Nonpoint Pollution Risk in North Carolina,” and Chapter 4: “A Matter of Scale: The Best Landscape Predictors of Water Quality Depend on Watershed Size.”)

Table 2.1: Variables used in analysis of stream invertebrate tolerance to stream degradation; the “Benthic Factors” column lists the impacts of the variable on macroinvertebrate tolerance, as described in Chen *et al.* 1993. The table lists the description of each variable included in the analysis, the scale at which it was analyzed (either the entire watershed; the area within 300-, 100-, and 50-foot riparian buffers; or both), the source of the data, the pathways through which it affects macroinvertebrate tolerance, and references to sources for further information on the variable and its relationship to nonpoint pollution.

Variable	Description	Scale	Data Source	Benthic Factors	References
Macroinvertebrate index scores (response)	1) N.C. Biotic Index (NCBI) 2) EPT Biotic Index (Ephemeroptera, Plecoptera, and Trichoptera)	NA	N.C. Division of Water Quality Biological Assessment Unit [1982-2000]	NA	Lenat 1993, Lenat 2002 (pers. comm.), NCDWQ 2001a
Land cover (predictor)	Percent of watershed or buffer that is: 1) Forested 2) Agricultural 3) Developed	watershed and buffer	Multi-Resolution Land Cover (MRLC) GIS raster data [1992]	flow regime, water quality, energy/food source, habitat structure	Basnyat <i>et al.</i> 1999, Diamond and Serveiss 2001, Genito <i>et al.</i> 2002, Lenat and Crawford 1994, Omernick <i>et al.</i> 1981, Sliva and Williams 2001, Sponseller <i>et al.</i> 2001, Wang <i>et al.</i> 1997, 2001
Topographic complexity (predictor)	Standard deviation of elevation of watershed or stream buffer	watershed and buffer	USEPA stream reach GIS data and USGS digital elevation model (DEM)	flow regime, water quality, habitat structure	Phillips 1989, Richards <i>et al.</i> 1996, Riekerk <i>et al.</i> 1988
Mean elevation (predictor)	Mean elevation of watershed or stream buffer	watershed and buffer	USEPA stream reach GIS data and USGS digital elevation model (DEM)	flow regime, water quality	Sponseller <i>et al.</i> 2001
Clay content of soil (predictor)	Percent of watershed or stream buffer with soil having at least 25 percent clay content	watershed and buffer	USGS State Soil and Geographic (STATSGO) GIS database	water quality, habitat structure	Cooper <i>et al.</i> 1987, Richards <i>et al.</i> 1996, Sliva and Williams 2001
Precipitation (predictor)	Amount of rainfall during the three months before stream sample	watershed	NOAA National Weather Service data [1982-2000]	flow regime, water quality	Brooks <i>et al.</i> 1991, Lenat 2002 (pers. comm.), Ward and Elliot 1995
Ecoregion (predictor)	Location in the Coastal Plain, Piedmont, and Southern Appalachians (dummy variable)	watershed	USEPA Ecoregions III GIS data	flow regime, water quality, habitat structure	NCDWQ 2001a Riekerk <i>et al.</i> 1988
Watershed area (predictor)	Watershed area (km ²)	watershed	USEPA stream reach GIS data and USGS digital elevation model (DEM)	flow regime, habitat structure, water quality	Brooks <i>et al.</i> 1991, Richards <i>et al.</i> 1996, Ward and Elliot 1995
Watershed shape (predictor)	Horton's Form Factor (area/square of watershed length)	watershed	USEPA stream reach GIS data and USGS digital elevation model (DEM)	flow regime, water quality	Brooks <i>et al.</i> 1991, Ward and Elliot 1995
Watershed slope/relief ratio (predictor)	Watershed elevation change/ watershed length from outlet to highest point on perimeter	watershed	USEPA stream reach GIS data and USGS digital elevation model (DEM)	flow regime, water quality	Brooks <i>et al.</i> 1991, Riekerk <i>et al.</i> 1988, Sliva and Williams 2001, Ward and Elliot 1995

c) Characterization of the potential and type of ecological effects that can be expected

I hypothesized that:

- (1) The vulnerability equations generated by stepwise regression analysis of landscape characteristics would predict a significant portion of the variability in benthic macroinvertebrate index scores.
- (2) Watersheds with more developed and agricultural land cover – both throughout the watershed, and within riparian zones – would have higher benthic macroinvertebrate index scores, or more stream degradation and a lower level of aquatic ecological integrity and water quality.
- (3) Watersheds with more forested land cover – throughout the watershed and in riparian zones – would have lower benthic macroinvertebrate index scores, reflecting better aquatic ecological integrity and water quality.
- (4) The type of landscape characteristics significant in affecting index scores would vary among North Carolina's three ecoregions (Southern Appalachians, Piedmont, and Coastal Plain).
- (5) The importance of land-use in riparian zones would differ between the largest and smallest watersheds in the study.

I used stepwise multiple regression (Draper and Smith 1998) to analyze the proportion of variability in the stream invertebrate tolerance indices attributable to differences in land cover and land form characteristics. I conducted identical regression analyses for the watershed and riparian-zone variables.

First, I tested the distribution of each of the 11 landscape variables and the two invertebrate index variables, and transformed several of the variables – with log or power transformations – to achieve a more normal distribution (SAS Institute Inc. 2000). Next, as discussed above, I ran simple linear regressions for each of the landscape variables against the invertebrate index variables, to determine the strength and direction – whether positive or negative – between the predictor and response variables.

Before running regressions on multiple variables, I tested the multicollinearity of the transformed landscape variables using Pearson correlation analysis (SAS Institute Inc. 2000). When two or more variables were correlated at a level of 0.8 or higher (O’Sullivan and Rassel 1999), I removed one or more variables so that only one remained. Relief ratio and average elevation were removed from analyses because both highly correlated with topographic complexity, which remained. Percent agricultural land cover was very highly correlated with percent forested, and was removed from both watershed and buffer multiple regression analyses; percent forested was included in the analyses. The Southern Appalachian location categorical variable was removed from the buffer analyses because it was correlated with Coastal Plain location and Piedmont location, both of which remained in the analyses.

During the stepwise regression of the multiple variables, I used an α -level of 0.05 for the significance level for the partial F entry test, and I used the conservative α -level of 0.15

for the partial F removal test (Draper and Smith 1998). I ran multiple regressions in SAS (SAS Institute Inc. 2000) with variables derived at the watershed level, and at the 300-foot, 100-foot, and 50-foot riparian zone widths. A handful of variables – ecoregion, rainfall, watershed area, and watershed shape – were included in both the watershed and riparian regressions.

The resulting coefficient of multiple determination (R^2) for each equation indicates the proportion of variability in the response variables – the invertebrate tolerance indices – attributable to the landscape variables included in the model. The unstandardized regression coefficients for each landscape variable represent that variable's weight and direction in the vulnerability index equation. The standardized (beta weight) regression coefficients (the result of rescaling each variable so that its mean is 0, and its standard deviation is 1) indicate the relative importance of the landscape characteristic relative to the other landscape variables in the model equation (O'Sullivan and Rassel 1999).

2.3.3 Phase 3: Risk Characterization

The third phase of ecological risk assessment – risk characterization – incorporates two steps: (a) summarizing assumptions, scientific uncertainties, and strengths and limitations of the analysis, and (b) producing a risk description that presents results of the process, including interpretation of any ecological threat, descriptions of uncertainty, and lines of evidence.

a) Summary of assumptions, scientific uncertainties, and strengths and limitations of analysis

1) *Uncertainties, and Strengths and Limitations of Analysis*

For a discussion of uncertainties and the strengths and limitations of the analysis, see Chapter 5: “Conclusions, Uncertainties, and Future Work.”

2) *Analysis Assumptions*

This study analyzes how landscape attributes at two scales – the catchment above a water quality sampling site, and three different riparian buffer widths – relate to benthic macroinvertebrate index scores for the sampling site. Like all risk assessments, it required a number of simplifying assumptions and analytical choices (Lackey 1997a). To describe the assumptions inherent in the ecological risk assessment, I outlined how I compiled the variables below. Assumptions important in the regression analysis itself are explained above in the Risk Analysis section titled “*Characterization of the potential and type of ecological effects that can be expected.*”

i) Watershed Delineation

For this study, I assembled the most complete list possible of NCDWQ benthic macroinvertebrate sampling sites and scores from across North Carolina. This information, for 2,000 sampling sites, was available through the Biological Assessment Unit’s Web site (N.C. Division of Water Quality 2002a), and from the latest basinwide water quality assessment reports (N.C. Division of Water Quality 1999a, 2000a, 2000b, 2000c, 2000d,

2000e, 2000f, 2001b, 2001c, 2002b). I shortened this list by eliminating locations that had not been sampled at least twice at roughly five-year increments between 1983 and 2001, from the period of May to September. I also excluded the sites that did not have both NCBI and EPT Biotic Index scores, and eliminated sites near the mouths of large river systems because the analysis focused on mostly headwaters watersheds smaller than 2,500 km². The result was a collection of 252 sites (Figure 2.4), many of which are regularly sampled as part of the NCDWQ's basinwide water quality assessment program.

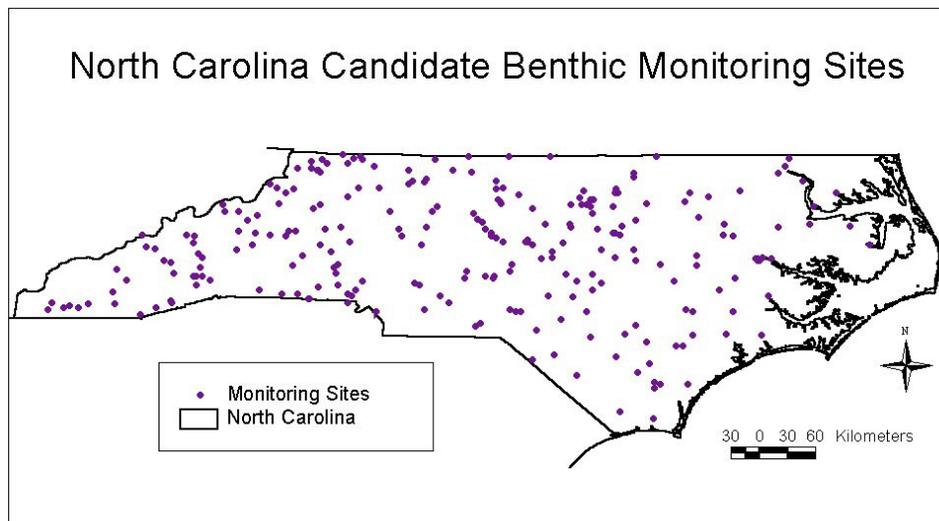


Figure 2.4: 252 North Carolina Division of Water Quality benthic macroinvertebrate monitoring locations

I narrowed the list again – to 125 sampling sites – by focusing only on sites from which invertebrate samples were taken in about 1992 and about 1997, because my initial goal was to study the relationships between changes in land cover over time (e.g., urban development and deforestation) with changes in water quality tolerance measures. However, adequately compatible geographic information system (GIS) land cover data did not exist at those two, or any other, intervals of time. In arriving at these 125 benthic monitoring sites, I

also excluded watersheds that extended into other states because the land cover data used in the analysis process was limited to North Carolina.

I next delineated the catchment associated with each of the 125 monitoring sites using the Geographic Information Systems (GIS) program ArcView 3.2 (ESRI 1999). I created a polygon shapefile for the boundary of each watershed using a NCDWQ-generated GIS point data set of North Carolina benthic monitoring sites (N.C. Division of Water Quality 2000g), along with a series of U.S. Environmental Protection Agency stream reach data sets (U.S. Environmental Protection Agency 1998d) and 300-meter resolution digital elevation models (U.S. Environmental Protection Agency 1998e) at the 8-digit U.S. Geological Survey (USGS) Hydrologic Unit level. Starting at the monitoring point, I followed the ridgelines around the stream segments above that site. When the boundaries of a catchment corresponded with the edge of USGS 8-digit Hydrologic Unit (U.S. Environmental Protection Agency 1998c), I followed that boundary in the creation of the catchment shapefile (Figure 2.5).

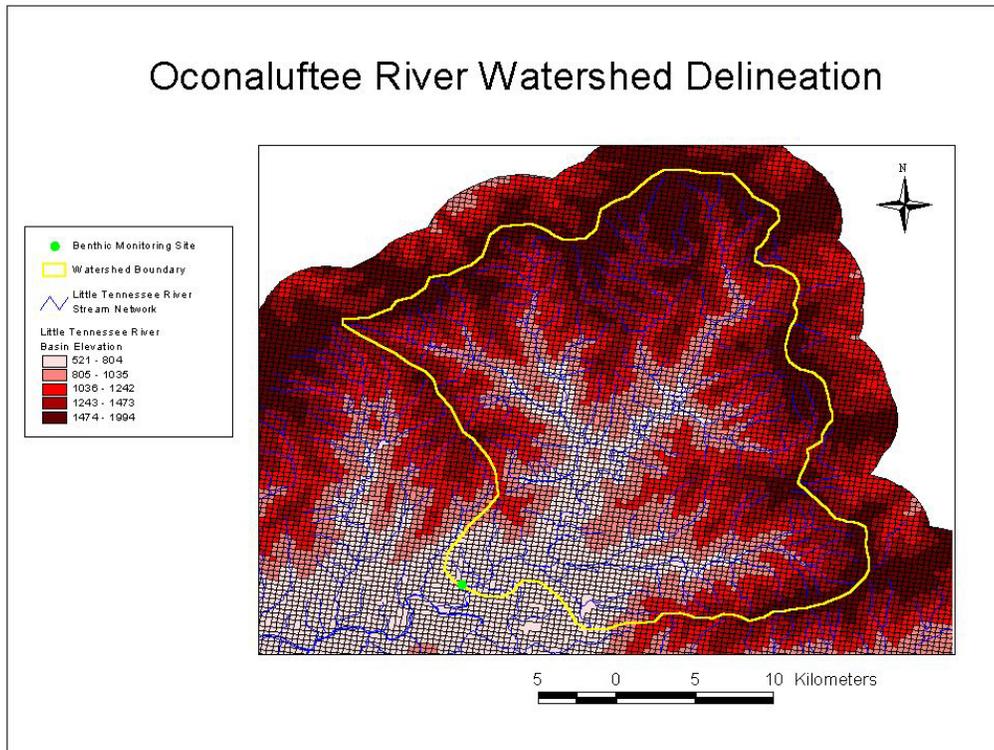


Figure 2.5: An example of watershed delineation on the Oconaluftee River, in the Little Tennessee River basin and the Southern Appalachian ecoregion

The following steps eliminated the largest catchments and randomly selected among catchments with overlapping boundaries. The 12 watersheds larger than 2,500 km² were excluded from the analysis because of a concern that they would overwhelm the effects of the landscape variables occurring in the smaller watersheds (which average 515.8 km² without the 12 largest watersheds), and because they skewed the otherwise normal distribution of the watershed size variable (when *log* transformed). The eliminated watersheds ranged in size from 2,920 km² to 7,721 km². It seemed appropriate to make this cut at 2,500 km² because of the large difference in the data between the smallest of the watersheds eliminated (2,920 km²) and the largest of the watersheds retained (2,374 km²).

Of the remaining 113 watersheds, 80 shared an area with other delineated watersheds; in other words, each of those 80 catchments included, or was enclosed by, another watershed. I assigned each of those watersheds a number from 1 to 80, and then used Matlab (The MathWorks, Inc. 2000) to generate the same numbers in random order. I then determined whether to retain or eliminate each watershed based on its placement in that new list; watersheds earlier in the list were included and those later in the list that overlapped were excluded.

The final result was 74 watersheds covering 38,168.5 km², or 30.25 percent of North Carolina's land area (Figure 2.6). Of these, 24 had their outlet in the Southern Appalachians ecoregion, 30 were in the Piedmont, and 20 in the Coastal Plain. The upper Piedmont and the lower Coastal Plain were somewhat underrepresented, for at least three reasons: (1) the NCDWQ's repeated sampling sites were not uniformly distributed across the state; (2) watersheds in the upper Piedmont and northern Coastal Plain often contained parts of Virginia, and (3) sites near the mouths of larger rivers such as the Roanoke and the Neuse were not considered because of the size of the watersheds that would have resulted.

These 74 catchment shapefiles were later used to compile several of the landscape variables included in the analysis of variation in the macroinvertebrate indices, including watershed size, watershed shape, watershed and stream buffer land cover, watershed and stream buffer topographic roughness, watershed and stream buffer minimum soil clay content, and the ecoregion of the sampling site.

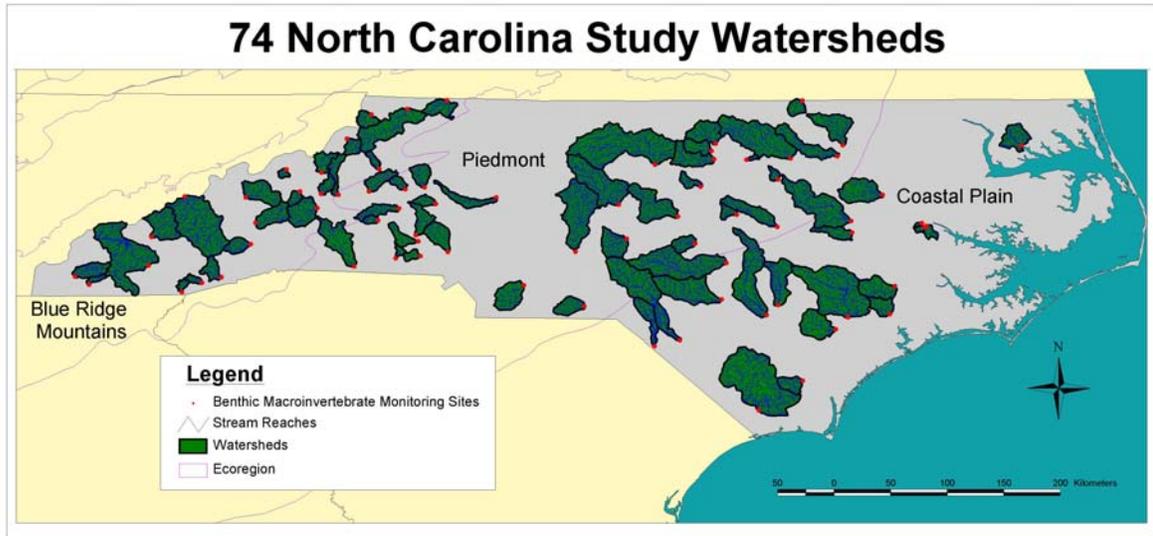


Figure 2.6: The 74 North Carolina project watersheds

ii) Creation of Stream Buffers

The project next required delineating 50-foot, 100-foot, and 300-foot riparian buffers along the streams in all 74 watersheds (Figures 2.7-2.10), using an approach similar to the one used by Dodd *et al.* (1994) to complete a buffering analysis for rivers and streams in the catchment of North Carolina's Albemarle-Pamlico estuary. To do this, I used ArcView 3.2's Geoprocessing Wizard (ESRI 1999), after clipping the EPA's stream reach layer (U.S. Environmental Protection Agency 1998d) for each catchment.

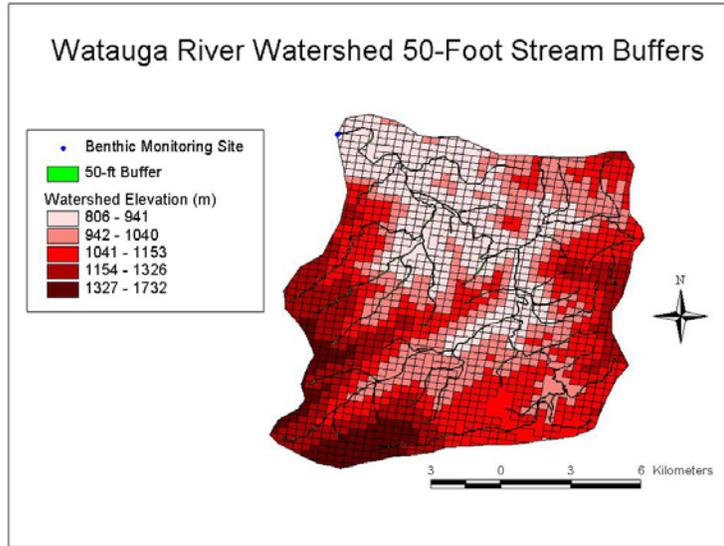


Figure 2.7: An example of 50-foot riparian buffer delineation on the Watauga River, in the Southern Appalachian ecoregion and the Watauga River basin

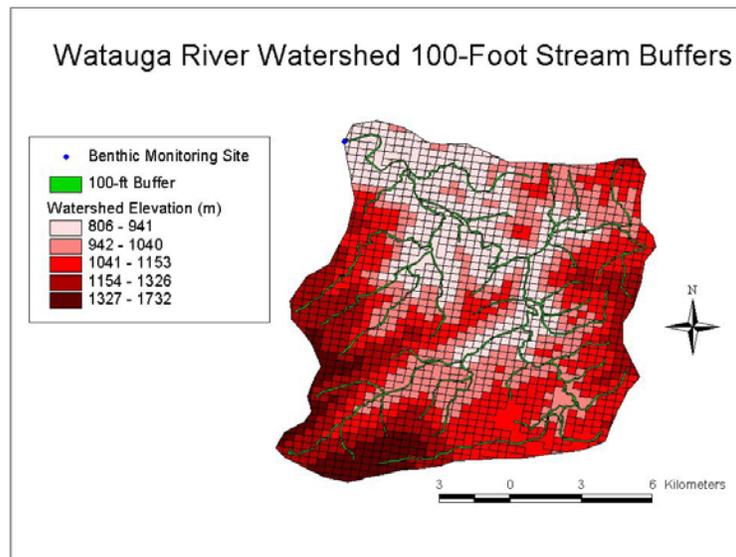


Figure 2.8: An example of 100-foot riparian buffer delineation on the Watauga River, in the Southern Appalachian ecoregion and the Watauga River basin

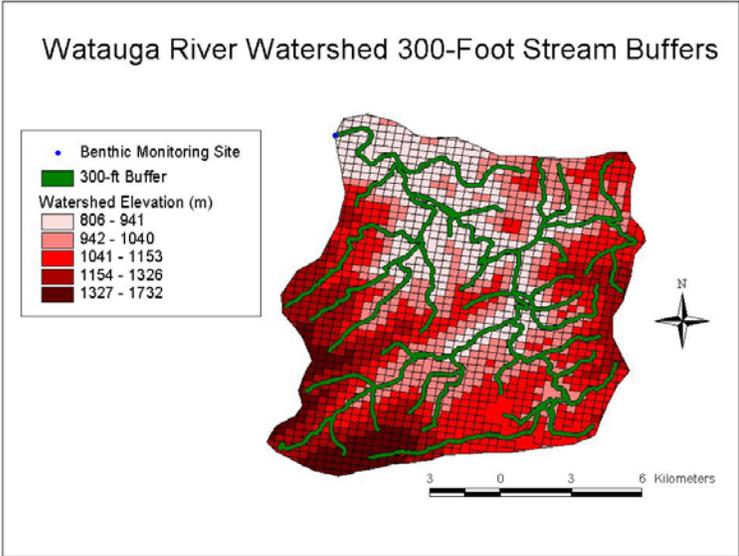


Figure 2.9: An example of 300-foot riparian buffer delineation on the Watauga River, in the Southern Appalachian ecoregion and the Watauga River basin

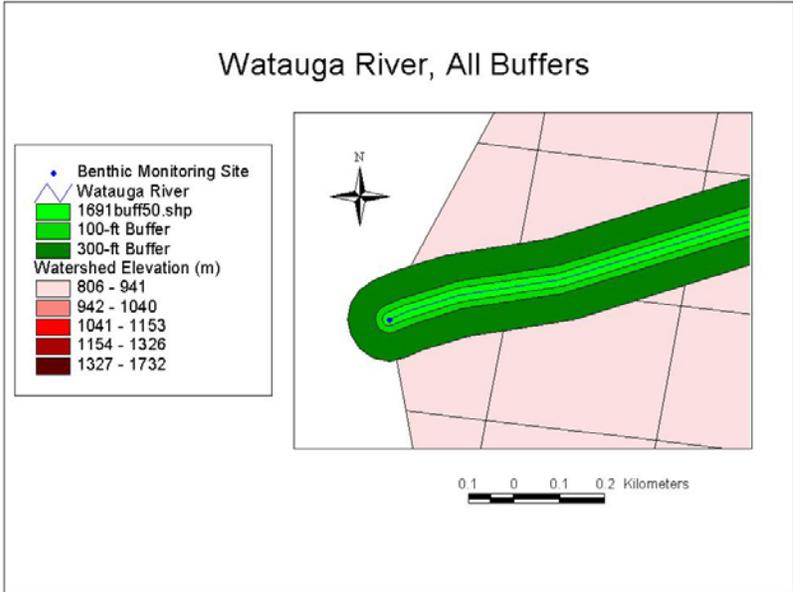


Figure 2.10: A close-up view of 50-foot, 100-foot, and 300-foot riparian buffers on the Watauga River

The 50-foot buffer was selected to reflect North Carolina’s riparian zone protection regulation, enacted permanently in 2000, to improve water quality in the Neuse River basin and the Tar-Pamlico River basin (N.C. Division of Water Quality 2000h; 15A North Carolina

Administrative Code 2B .0233, .0241-.0242, .0259-.0261), and enacted temporarily for portions of the Catawba River basin in 2001 (N.C. Division of Water Quality 2001d; 15A North Carolina Administrative Code 2B .0243-.0244). A 100-foot protected buffer along streams is considered by many adequate to protect water quality from sediment and other pollutants (Dr. James Gregory, pers. com.; sources referenced in Large and Petts 1994), while a 300-foot buffer is almost universally accepted as adequate to protect stream water quality (Richards *et al.* 1996, Large and Petts 1994).

iii) Landscape Characteristics

To investigate variability in stream invertebrate biotic indices, I assembled 11 landscape predictor variables for each watershed (Table 2.1). Of those, six were assembled at both the watershed-scale and each of the three buffer widths (percent forested, percent agricultural, percent developed, topographic complexity, mean elevation, and soil clay content). The remaining five were assembled only at the watershed scale (precipitation, watershed area, Horton's form factor, relief ratio, and ecoregion).

Land cover variables

Three land cover variables were derived at both the watershed and buffer scales – percent forested, percent agricultural, and percent developed – using a raster grid GIS layer from 1992. This information (Figure 2.11), commissioned by the Multi-Resolution Land Characterization (MRLC) Consortium, is 30-meter Landsat thematic mapper (TM) raster data for the state of North Carolina (Multi-Resolution Land Characterization Consortium

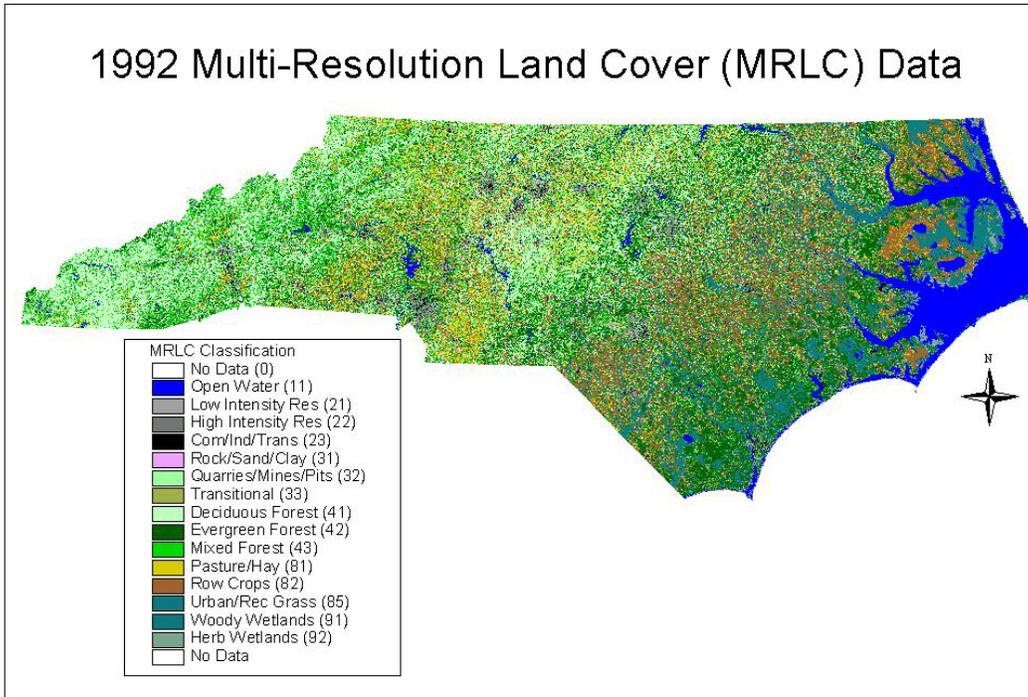


Figure 2.11: 1992 Multi-Resolution Land Characterization (MRLC) Consortium National Land Cover Data for North Carolina

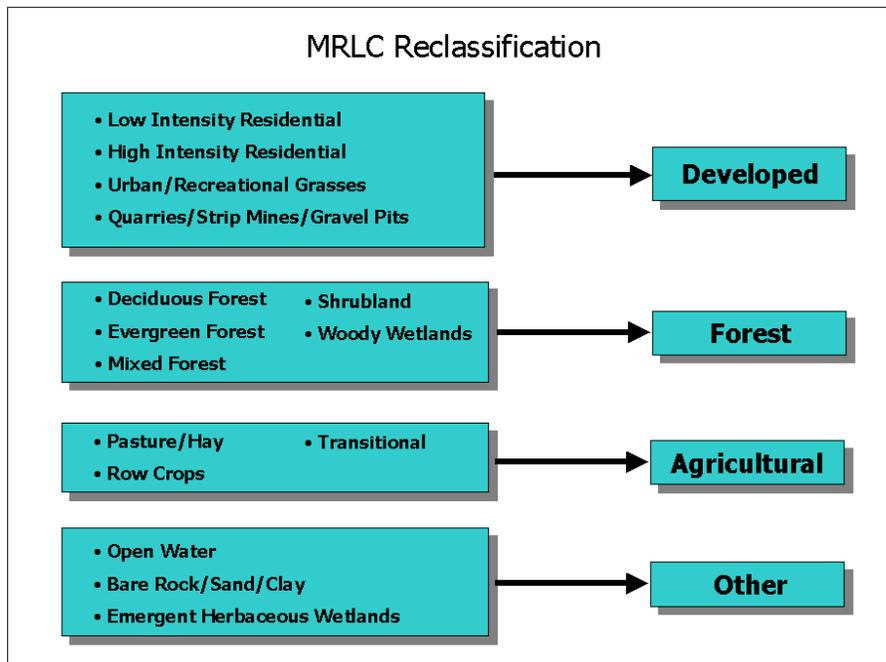


Figure 2.12: Reclassification of 1992 Multi-Resolution Land Characterization (MRLC) National Land Cover Data for North Carolina

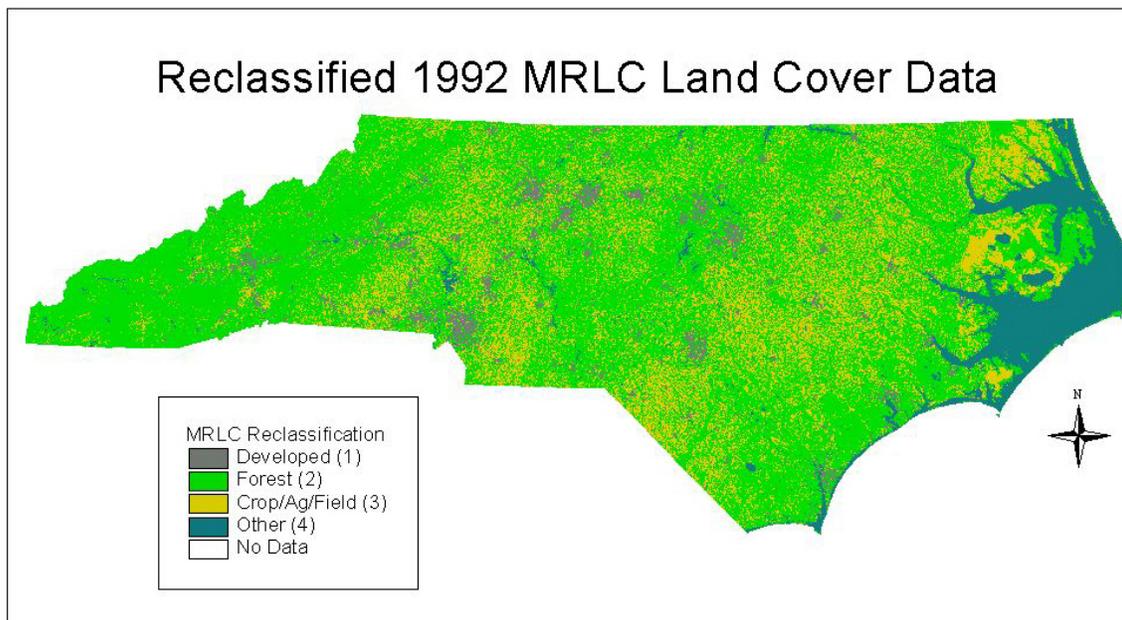


Figure 2.13: 1992 Multi-Resolution Land Characterization (MRLC) Consortium National Land Cover Data for North Carolina, aggregated into four classifications

2000). It consists of 15 data classifications for North Carolina, which I reclassified into four: developed, forest, agricultural, and other (Figure 2.12-2.13).

The reclassified MRLC land cover data and the watershed and stream buffers were all projected to a common coordinate system – North Carolina State Plane, North American Data (NAD) 1983 – using ArcGIS 8.1 (ESRI 2001). For each catchment, the single watershed shapefile and the three stream buffer shapefiles (50-foot, 100-foot, and 300-foot) were converted to grids using ArcView 3.2 (ESRI 1999), and were then used to mask the statewide land cover data. Next, for each watershed and buffer I added the number of 30-meter cells with forest, agricultural, or developed classifications, and used that number to find the relative percent of the watershed or buffer in each of the three land cover types. This process eliminated land uses with a classification of “other” from the analysis.

Land form characteristics

This analysis involved computing three land form characteristics for each watershed – watershed area, shape, and relief – and two land form characteristics for both watersheds and stream buffers – topographic complexity and mean elevation.

Watershed area, in square kilometers, was computed by ArcView 3.2 (ESRI 1999) following the delineation of each of the 74 catchments. The mean area was 515.8 km². The relief ratio for the catchment was determined by finding the highest point on the perimeter of the watershed Digital Elevation Model and measuring the distance to the mouth of the watershed. The relief ratio is calculated with the following equation (Gregory and Walling 1973):

$$R = \frac{\Delta E}{L}$$

where:

R is the relief ratio for the watershed,

ΔE is the difference in elevation, in meters, between the high point on the perimeter of the watershed and the mouth of the watershed, and

L is the length of the watershed, in kilometers, between the high point and the mouth.

Calculating the area of the watershed and then measuring the distance between its mouth and farthest point allowed for the quantification of the catchment's shape, using Horton's Form Factor (Gregory and Walling 1973):

$$R_f = \frac{A}{L^2}$$

where:

R_f is Horton's Form Factor,

A is the area of the watershed, in km^2 , and

L^2 is the squared length of the watershed, in kilometers, between the mouth and the most distant point on the perimeter.

Smaller Horton Form Factor scores indicate watersheds that are elongated and, as a result, tend to have slower precipitation drainage and therefore less erosion. Higher scores mean a watershed is more compact and may have quicker drainage and more erosion. More erosion can negatively impact stream invertebrate communities by degrading habitat, by filling of spaces in the stream bottom matrix with sediment (Richards and Host 1994), and by transporting pollutants (Neary *et al.* 1988).

The mean elevation for each catchment or buffer was determined in ArcView by calculating the mean elevation, in meters, of the Digital Elevation Model vector elevation units (Figure 2.14-2.15). Topographic complexity for each catchment or buffer was found in the same fashion, but by calculating the elevation standard deviation.

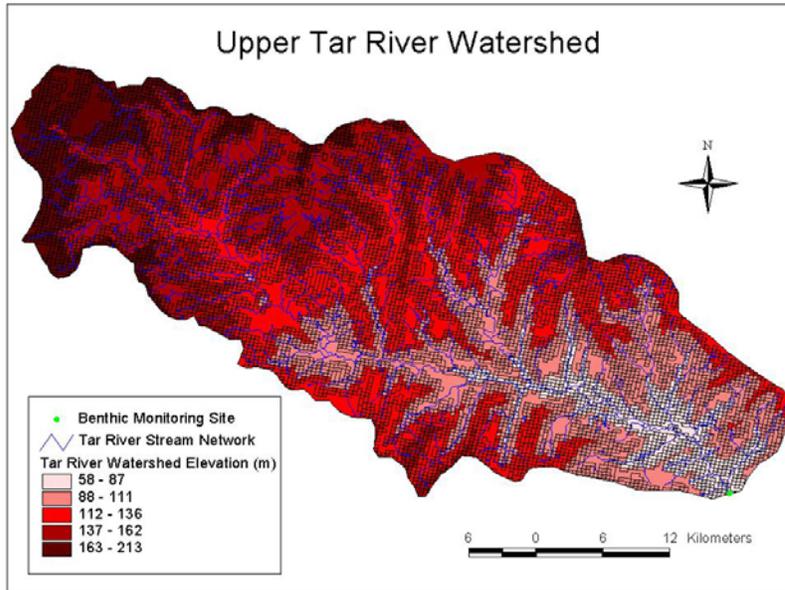


Figure 2.14: The mean elevation and topographic complexity were found for each watershed (including the Upper Tar River watershed, shown here) by calculating the mean elevation, in meters, and standard deviation of the Digital Elevation Model vector elevation units.

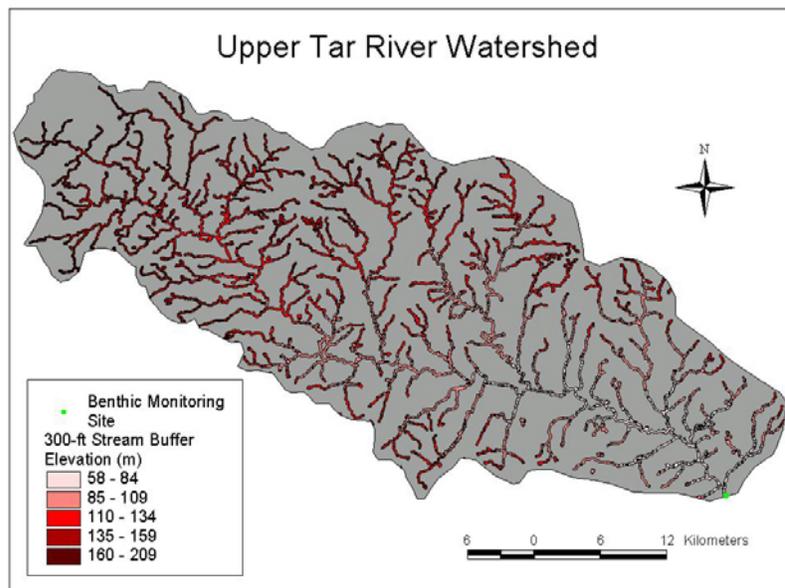


Figure 2.15: The mean elevation and topographic complexity were found for each buffer width (including the 300-foot width in the Upper Tar River watershed, shown here) by calculating the mean elevation, in meters, and standard deviation of the Digital Elevation Model vector elevation units.

Ecoregion

The ecoregion of a sampling site has a significant impact on the types of invertebrates present in a North Carolina stream or river (N.C. Division of Water Quality 2001a, Riekerk et al. 1988). Additionally, the Biological Assessment Unit in the N.C. Division of Water Quality (2001a) has found that invertebrate tolerance ratings that qualify as “excellent,” “good,” “fair,” or “poor” differ by ecoregion. For example, an “excellent” score is one below 4.05 in the Southern Appalachians, below 5.19 in the Piedmont, and below 5.47 in the Coastal Plain. The EPA Level III Ecoregion of each sampling site (U.S. Environmental Protection Agency 1998c), therefore, was included as a “dummy” categorical variable (Draper and Smith 1998) in the analysis (Figure 2.6).

Rainfall

Precipitation can have an important impact on the assemblage of macroinvertebrates inhabiting a stream or river by influencing both the flow regime and water quality of the stream or river (Caruso 2002; Caruso 2001; Feminella 1996). Therefore, this analysis includes as an independent variable the amount of precipitation that fell in the three months preceding each stream invertebrate sample. This data was taken from a summary of monthly precipitation totals assembled by the National Climatic Data Center (U.S. Department of Commerce 1999). The precipitation data used were from the nearest weather monitoring station within the catchment for the invertebrate sampling site (Figure 2.16). If no weather station was located in the catchment, data were taken from the nearest station outside the watershed but above the sampling site, when possible. If data were not available for a given time and place, they were taken from the next nearest station. Twenty-seven of the 74

catchments (36.5 percent) contained the rainfall collection station from which the precipitation data were taken; the mean distance was 15.75 kilometers between each benthic monitoring site and its associated weather station.

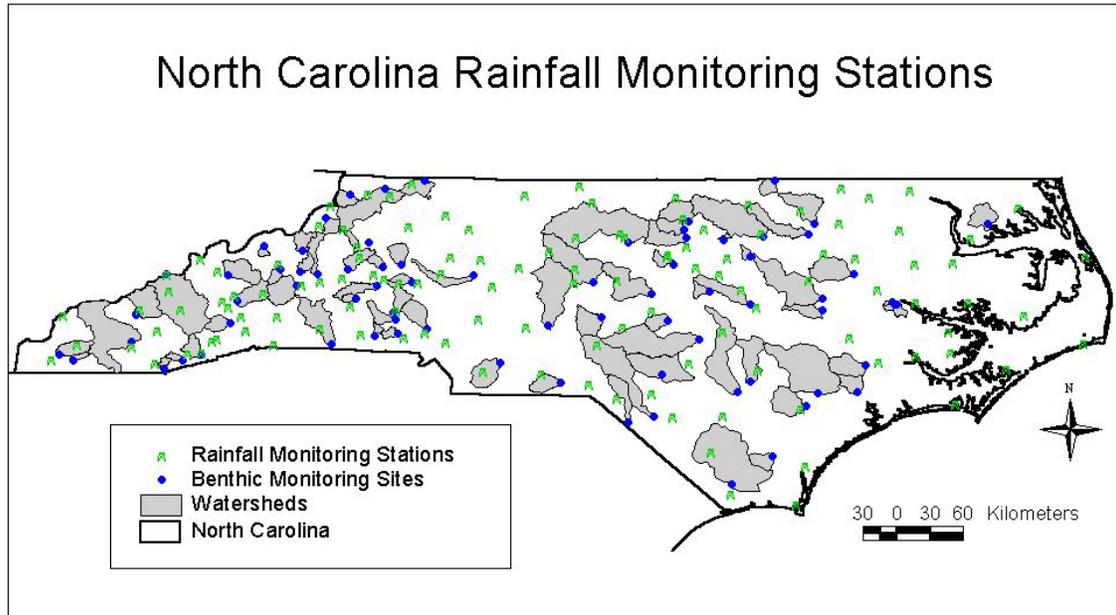


Figure 2.16: Precipitation data were taken from the rainfall monitoring stations nearest to the benthic sampling sites, upstream from the sampling site if possible.

Soil clay content

Clay content in soil is associated with increased sedimentation and chemical pollution in streams and rivers (Cooper *et al.* 1987, Richards *et al.* 1996, Sliva and Williams 2001), which can impact the structure of macrobenthic invertebrate communities (Sliva and Williams 2001, Richards *et al.* 1996). The minimum soil clay content of watersheds and buffers, therefore, was included as a variable in this analysis – specifically, the percent of the catchment or buffer zone area having soils that contain at least 25 percent clay content (Figure 2.17-2.18). I assembled this information in ArcView for each watershed and buffer

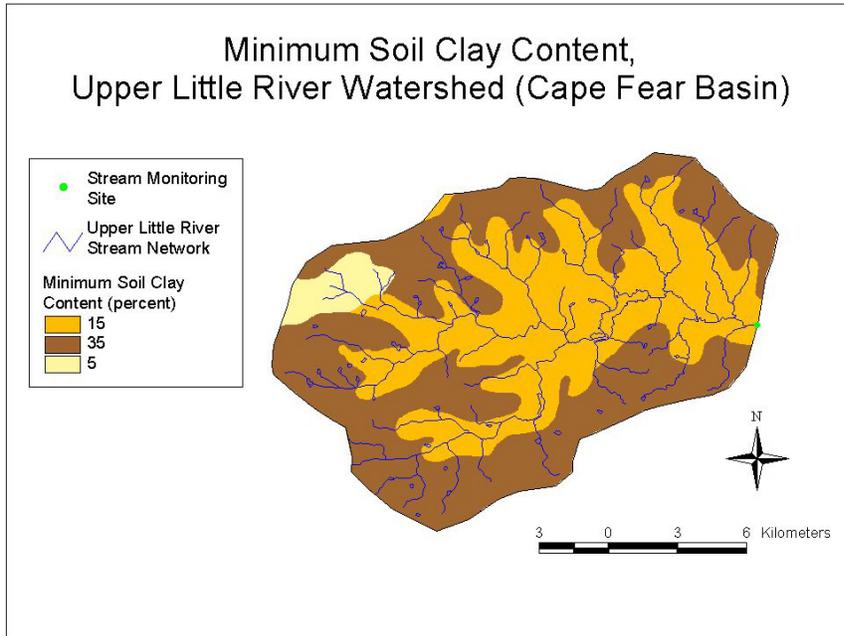


Figure 2.17: Minimum soil clay content of the Upper Little River watershed, in the Cape Fear Basin

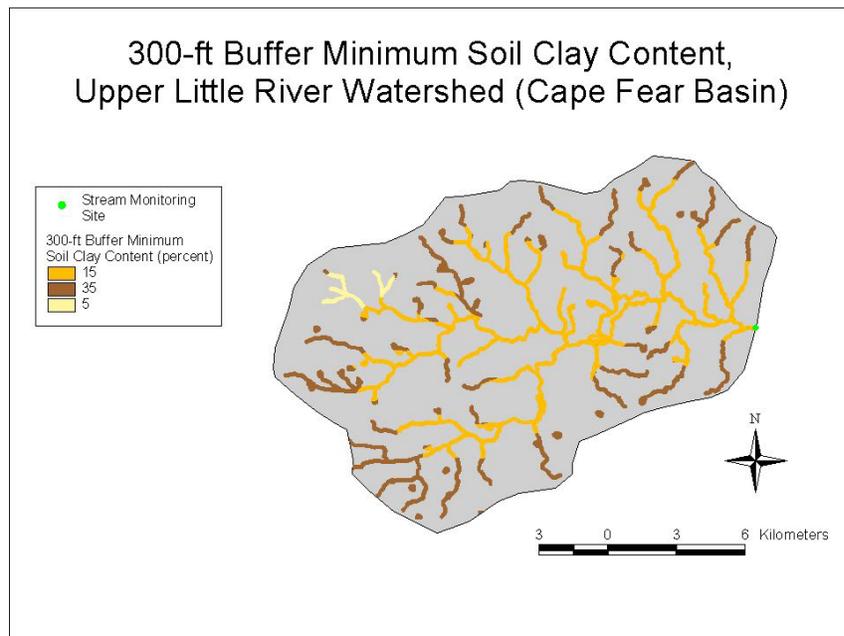


Figure 2.18: Minimum soil clay content of 300-foot riparian zones of the Upper Little River watershed, in the Cape Fear Basin

shapefile by clipping minimum soil clay content data from the U.S. Geological Survey's State Soil and Geographic (STATSGO) GIS database (Schwarz and Alexander 2000).

b) Production of risk description that presents results of the process

For an overview of the risk description resulting from the ecological risk assessment process, see Chapter 3: "A Watershed-Scale Model for Predicting Nonpoint Pollution Risk in North Carolina," and Chapter 4: "A Matter of Scale: The Best Landscape Predictors of Water Quality Depend on Watershed Size."

2.3.4 Phase 4: Communication of Assessment Results

The results of this project were presented in a thesis seminar at the North Carolina State University Department of Forestry on August 20, 2002. Additionally, the results were presented at the joint Ecological Society of America/Society for Restoration Ecology national conference August 5, 2002, in Tucson, Arizona. Chapter 3 and Chapter 4 of this thesis will be submitted for publication in one or more appropriate peer-reviewed forestry, landscape ecology, or water quality journals.

Chapter 3

A Watershed-Scale Model for Predicting Nonpoint Pollution Risk in North Carolina

Prepared for submission to *Environmental Management*

3.1 Abstract

The Southeastern United States is a global center of freshwater biotic diversity, but much of the region's aquatic biodiversity is at risk from stream degradation. Nonpoint pollution sources are responsible for 70 percent of that degradation, and controlling nonpoint pollution from agriculture and urbanization is considered critical to maintaining future water quality and aquatic biodiversity in the Southeast. We used an ecological risk assessment framework to develop vulnerability models that can help policymakers and natural resource managers understand the impact of land cover changes on water quality and aquatic ecological integrity in North Carolina. Additionally, we determined which landscape characteristics are most closely associated with macroinvertebrate community tolerance of stream degradation, and therefore with lower-quality water. The results will allow managers and policymakers to weigh the risks of management and policy decisions to a given watershed or set of watersheds, including whether streamside buffer protection zones are ecologically effective and economically efficient in achieving water quality standards. Regression analysis revealed that landscape variables explained up to 77 percent of the variability in benthic macroinvertebrate index scores. The resulting vulnerability models indicate that North Carolina watersheds with flatter topography and increasing urban development are most at risk for degraded water quality and stream habitat conditions. A useful approach for characterizing the risk of potential land management options is to use the vulnerability models to "simulate" land use activities, such as conversion of land cover.

3.2 Introduction

The Southeastern United States is a region with extensive forestlands and high-quality aquatic resources, despite the extensive loss of forest and the degradation of water quality since European settlement (West 2002). In a nation that is a global center of freshwater biotic diversity, the rivers and streams of the Southeast harbor an extraordinary variety of life, resulting from diverse physical geography, a favorable climate, and a dynamic natural history (Chaplin *et al.* 2000). Ninety-one percent of the world's freshwater mussel species and more than half the known fingernail clam and snail species occur in the Southeast (Neves *et al.* 1997), and the region is home to half the 800 fish species native to the United States and Canada (Warren *et al.* 1997). The region is home to 864 rare aquatic species distributed among seven taxonomic groups – fish, mussels, snails, insects, crustaceans, reptiles, and amphibians (Herrig and Shulte 2002). The preservation of this aquatic biodiversity will require maintaining and, in some cases, improving the conditions necessary for the survival of a wide variety of species.

Aquatic biodiversity in the Southeast is at risk from a variety of pollutants. In 1998, 45 percent of the total river miles assessed by Southeastern states – totaling 103,441 miles – was classified as impaired by some form of pollution or habitat degradation. That number increased from the 26 percent of river miles listed as impaired in 1988 (West 2002). Between 1988 and 1998, 70 percent of stream degradation in the Southeast resulted from nonpoint source pollutants, those not traceable to a discrete facility. Controlling nonpoint pollution from its most significant sources – urban development and agriculture – is considered critical to maintaining future water quality in the Southeast (West 2002). Nonpoint pollution can have negative consequences for the integrity of aquatic biota by

altering physical habitat, modifying seasonal water flow, altering the systemic food base, contaminating water with toxic chemicals, and modifying interactions among organisms (Karr 1999, Chen 1994).

Change in land use affects water quality and aquatic biodiversity through increased nonpoint pollution (Hunsaker *et al.* 1992). It is difficult, however, to separate the effects of land use from those associated with other landscape characteristics. The physical characteristics of streams, which are closely related to aquatic ecological integrity, are influenced by a wide variety of landscape features (Richards *et al.* 1996). To characterize the ecological condition of streams, it is important to assess the relative risk human activities pose to aquatic ecosystems. That requires assessing the relative risk of a variety of human activities by placing sample sites on a human disturbance gradient, to allow for the interpretation biological responses and the separation of human disturbance from natural conditions (Bryce *et al.* 1999).

Ecological risk assessment holds promise as a tool for comparing the relative risks of human activities to water quality (Zandbergen 1998, Chen *et al.* 1993). It is a process used to systematically evaluate and organize data, information, assumptions, and uncertainties to better understand and predict the relationships between stressors and ecological effects in a fashion useful for environmental decision-making (U.S. Environmental Protection Agency 1998a). Risk assessment requires the use of formal quantitative techniques to estimate the probabilities of effects on well-defined endpoints, and to estimate uncertainties associated with the analysis. It is also clearly separated from the process of choosing among alternatives and determining the acceptability of risks, which is the responsibility of policymakers (Suter 1993).

The resident biota of aquatic ecosystems serve as continuous monitors of cumulative effects on those systems, and are often used as endpoints in ecological risk assessments (Diamond and Serveiss 2001, Bryce *et al.* 1999, Zandbergen 1998, U.S. Environmental Protection Agency 1990). Specifically, the macrobenthic invertebrate organisms associated with the substrates of rivers, streams, and lakes, are considered especially good indicators of water quality (Karr 1999, Metcalfe-Smith 1994, Lenat 1993, Plafkin *et al.* 1989, Hilsenhoff 1987). These include the aquatic nymphs and larvae of such insect orders as Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies). Because macrobenthic invertebrates are less mobile than other types of aquatic organisms, their often-diverse communities respond to a wide variety of pollutants. These communities can reflect both short-term and long-term trends because many species have life cycles longer than a year (N.C. Department of Water Quality 2001a, Metcalfe-Smith 1994). Additionally, macrobenthic invertebrate community structure reflects nonpoint source pollution from silvicultural activities because these organisms are abundant in low-order streams where much timber harvesting occurs, and because they are sensitive to habitat and water quality changes (Adams *et al.* 1995).

Several researchers have detected relationships between land-use and macroinvertebrate community composition. Lenat and Crawford (1994) concluded that land use in North Carolina Piedmont watersheds strongly influenced invertebrate communities. The forested stream in their study had high invertebrate taxa richness, a low biotic index, and many unique species, resulting in a “good” water quality classification. Measurements of the same criteria indicted moderate stress (fair water quality) at an agricultural site and severe stress (poor water quality) at an urban site. The urban stream was characterized by low

species richness for most invertebrate groups and very low abundance values, while the agricultural site had highest abundance values, indicating stream nutrient enrichment. Research by Richards *et al.* (1996) in southwestern Wisconsin suggested that macroinvertebrate community assemblages were strongly influenced by land-use characteristics – positively by the amount of forested wetlands, and negatively by row-crop agriculture.

Other researchers have used multiple regression analysis of spatial data to determine the relationship between watershed landscape attributes and macrobenthic invertebrate community composition. Stepwise regression analysis by Sponseller *et al.* (2001) on data from the Roanoke River basin of southern Virginia indicated that local, upstream patches of riparian forest might have a positive effect on macroinvertebrate community structure. Harding *et al.* (1998), also using stepwise regression, found that invertebrate richness and measures of diversity were significantly greater in forested streams than agricultural streams in the Little Tennessee River and French Broad River basins of western North Carolina.

The current study follows the approach of Diamond and Serveiss (2001) and Hunsaker *et al.* (1992), who used stepwise regression as part of an ecological risk assessment of how land use affects water quality. Diamond and Serveiss examined how human land use and stream habitat quality impact native fish and mussel populations in the Clinch and Powell river basin of southwestern Virginia. Hunsaker *et al.* investigated the relationship between land-use characteristics and water purity, as measured by water electrical conductivity.

For this project, we developed vulnerability models to help policymakers and natural resource managers understand the impact of land cover changes on water quality in North

Carolina. Specifically, we determined which landscape characteristics at two different scales – the entire watershed, and riparian zones 30.48 meters (100 feet) along both sides of streams – are the best predictors of macroinvertebrate community integrity. That information, in turn, can be used to document spatial and temporal changes in water quality (N.C. Division of Water Quality 2001a). The results will allow managers and policymakers to weigh the risks of management and policy decisions on a given watershed or set of watersheds, including whether streamside buffer protection zones are ecologically effective and economically efficient in achieving water quality standards.

3.3 Methods

3.3.1 Landscape Variables

For this study, 11 land cover and land form characteristics (Table 3.1) were measured for 74 North Carolina watersheds (Figure 3.1). These variables were proportion forested, proportion agricultural, proportion developed, precipitation, watershed area, watershed shape, relief ratio, topographic complexity, mean elevation, soil clay content, and ecoregion. Of these, the three land cover variables were also measured for riparian zones 30.48 meters (100 feet) on either side of streams. We used ArcView 3.2 (ESRI 1999) to delineate the watersheds and create buffer files, and ArcGIS 8.1 (ESRI 2001) to project all the files to a common coordinate system – North Carolina State Plane, North American Data (NAD) 1983.

Three land cover variables were derived at both the watershed and buffer scales – percent forested, percent agricultural, and percent developed – using a raster grid GIS layer from 1992. This information, commissioned by the Multi-Resolution Land Characterization (MRLC) Consortium, is 30-meter Landsat thematic mapper (TM) raster data for the state of

North Carolina (Multi-Resolution Land Characterization Consortium 2000). It consists of 15 data classifications for North Carolina, which we reclassified into four: developed, forest, agricultural, and other (Figure 3.2). For each catchment, the single watershed files and the stream buffer files were converted to grids using ArcView 3.2 (ESRI 1999), and were then used to mask the statewide land cover data. For each watershed and buffer, we found the relative proportion of each land cover type by dividing the number of 30-meter cells in a given land cover type by the total number of cells in the forest, agricultural, and developed classifications. This process eliminated land uses with a classification of “other” from the analysis.

3.3.2 Macroinvertebrate Indices

We determined which of these landscape characteristics have a significant impact on the integrity of stream life, as measured by two metrics that describe the tolerance of benthic macroinvertebrates to stream degradation. These metrics are the North Carolina Biotic Index (NCBI); and an index of Ephemeroptera (mayfly), Plecoptera (stonefly), and Trichoptera (caddisfly) tolerance (Lenat 1993). The NCBI is considered a more reliable indicator of stream chemistry and habitat quality than of in-stream sediment (N.C. Division of Water Quality 2001a), which is among the more significant pollutant sources in the Southeast. The EPT Biotic Index (EPTBI) is believed to better measure the impact of sediment in streams (David Lenat, pers. com.), but is considered a less reliable measure of water quality, especially at high elevation sites and sites at which low numbers of EPT organisms are collected. Both indices are scored on a scale of 0 to 10, with 0 indicating the presence of stream invertebrates least tolerant of degradation (and therefore the existence of better water

quality), and 10 indicating an invertebrate community most tolerant of degradation (and therefore lower water quality).

For this study, we assembled a complete list of NCDWQ benthic macroinvertebrate sampling sites and scores from across North Carolina, available from the Biological Assessment Unit (N.C. Division of Water Quality 2002a) and from the latest basinwide water quality assessment reports (N.C. Division of Water Quality 1999a, 2000a, 2000b, 2000c, 2000d, 2000e, 2000f, 2001b, 2001c, 2002b). We narrowed the list of about 2,000 sampling sites by excluding the sites that have watersheds extending into neighboring states, and by eliminating sites at the mouths of watersheds larger than 2,500 km². We additionally focused only on sites from which both NCBI and EPT Biotic Index invertebrate samples were taken in about 1992 and about 1997, because our initial goal was to study the relationships between changes in land cover over time (e.g., urban development and deforestation) with changes in water quality tolerance measures. However, adequately compatible spatial land cover data did not exist at these two, or any other, intervals of time. Several of the remaining 125 watersheds were nested within other watersheds. When this was the case, one of the nested watersheds was chosen at random to be included among the 74 watersheds incorporated into the analysis.

3.3.3 Data Analyses

We transformed data as needed to achieve a normal distribution for each variable (SAS Institute Inc. 2000). We ran bivariate linear regressions to determine the strength and direction between the transformed landscape variables and the invertebrate index variables. A larger coefficient of determination (r^2) value indicated a stronger relationship between

variability in the landscape characteristic and the invertebrate index, while a positive coefficient suggested that a landscape variable is associated with invertebrate taxa tolerant of greater stream degradation. A positive regression coefficient, therefore, indicates the landscape variable has a negative correlation with water quality, while a negative coefficient has a positive effect.

Before running regressions on multiple variables, we tested the multicollinearity of the landscape variables using Pearson correlation analysis (SAS Institute Inc. 2000). When two or more variables were correlated at a level of 0.8 or higher (O'Sullivan and Russel 1999), we removed one or more variables so that only one remained. Relief ratio and average elevation were removed from analyses because both were highly correlated (>0.9) with topographic complexity, which remained. The proportion of agricultural land cover was almost the reverse of the proportion forested (correlation of -0.98), and was removed from both watershed and buffer multiple regression analyses; the forest variable was included in the analyses. The developed proportion of riparian zones was correlated with the developed proportion of the watershed (>0.8), and was dropped from consideration, as was the agricultural proportion of riparian zones, which was highly correlated with the proportion of the buffer that is forested. The Southern Appalachian location categorical variable was removed from the buffer analyses because it was correlated with Coastal Plain location and Piedmont location, both of which remained in the analyses.

Stepwise multiple regression (Draper and Smith 1998) was used to analyze the proportion of variability in the stream invertebrate tolerance indices attributable to differences in land cover and land form characteristics. We used an α -level of 0.05 as the significance level for the partial F entry test, and we used the conservative α -level of 0.15 for

the partial F removal test (Draper and Smith 1998). We ran multiple regressions in SAS (SAS Institute Inc. 2000) with variables derived at the watershed scale. The resulting coefficient of multiple determination (R^2) for each equation indicates the proportion of variability in the invertebrate tolerance indices attributable to the landscape variables included in the model. The unstandardized regression coefficients for each landscape variable represent that variable's weight and direction in the vulnerability index equation. The standardized (beta weight) regression coefficients indicate the relative importance of the landscape characteristic relative to the other landscape variables in the model equation.

3.4 Results

3.4.1 Bivariate Regression Results

The simple, bivariate regression analysis found statistically significant relationships between most of the landscape variables and the NCBI and EPTBI scores; the exceptions were watershed area, watershed shape, and soil clay content (Table 3.2).

Two of the three watershed land cover variables – proportion agricultural and proportion forested – exhibited somewhat strong relationships (Table 3.3). The proportion of agriculture at the watershed scale had a positive relationship with the indices, meaning that it was negatively correlated with aquatic ecological integrity. The proportion of forest was correlated with better stream conditions. For both land cover types, the relationship was strongest with the North Carolina Biotic Index. The proportion of developed land cover also had a statistically significant relationship with the indices, but the r^2 value was considerably lower. It showed a stronger relationship with the EPT Biotic Index, and was correlated with degraded stream conditions.

The land form feature exhibiting the strongest correlation with invertebrate indices was topographic complexity (Table 3.4). Mean elevation and relief ratio, which are closely related to topographic complexity, demonstrated relationships nearly as strong. These variables have a negative relationship with macrobenthic invertebrate tolerance to stream degradation, indicating that greater topographic complexity, relief ratio, and mean elevation are each associated with better water quality. Rainfall was the only other variable with a statistically significant relationship with macrobenthic invertebrate community structure. This correlation was also negative, meaning that greater amounts of rainfall were more likely to be accompanied by better water quality and aquatic habitat conditions.

3.4.2 Multiple Regression Results

The full regression models containing all 10 of the landscape variables explain between 71 percent and 77 percent of the variability in the benthic macroinvertebrate tolerance indices (Table 3.5). The risk assessment vulnerability models for the two indices are listed as the regression equations at the bottom of Tables 3.6 and 3.7.

For the North Carolina Biotic Index, the landscape attribute variables predicted 76.6 percent of the variation (Table 3.6). Topographic complexity, percent of watershed developed, and watershed shape were the most important variables in the model, as indicated by the beta weight (standardized regression coefficients) for each. The remaining variables were less important in the model and were considerably less statistically significant.

Topographic complexity was, again, the most important variable in the EPT Biotic Index regression model, followed by percent of watershed developed, and whether the sampling site was located in the Coastal Plain (Table 3.7).

3.4.3 Reduced Regression Models

Finding the “best” regression models for each index will reduce the time and costs model users have to spend collecting parameter data, and will keep the variance of the regression predictions small by limiting the number of parameters in the model (Draper and Smith 1998). The reduced models that resulted from using stepwise regression are listed at the bottom of Tables 3.9 and 3.10. They used only two or three landscape characteristics to predict between 69.6 percent and 74.7 percent of the variability in stream invertebrate index scores, depending on the index used (Table 3.8). Thus, these reduced models explained nearly all the variability explained by the 10-variable full models. More variability was explained in the North Carolina Biotic Index scores than in the EPT Biotic Index. Both models were highly significant.

One land form variable (topographic complexity) and one land cover variable (proportion of watershed developed) are the only two significant variables common to both indices (Tables 3.9-3.10). In both regression models, topographic complexity is positively correlated with better water quality. The opposite is true with the percent of watershed developed, which has a negative relationship with better water quality. Both of these relationships were expected. The only other significant variable – whether a sampling site is located in the Coastal Plain – occurs in the EPT index (Table 3.10). Like topographic complexity, this variable is positively correlated with better stream conditions, although the opposite (and less statistically significant) relationship occurred for this variable in the North Carolina Biotic Index (Table 3.9).

In both index models, topographic complexity was the relative strongest predictor of macroinvertebrate community assemblage, as noted by the standardized regression coefficients (beta weights). In other words, the more topographically complex a North Carolina watershed is, the more likely it is to have a benthic macroinvertebrate community assemblage indicative of higher-quality stream habitat. This holds true for the mean elevation and relief ratios of watersheds, since those variables are highly correlated with topographic complexity. The strength of the relationship between topographic complexity and water quality is somewhat reduced by the amount of developed area in a watershed, which is negatively associated with water quality.

3.4.4 Ecoregion Results

Reduced regression models for sites in North Carolina's three ecoregions generally resulted in lower R^2 values than did the statewide reduced models (Table 3.11). The Coastal Plain models were better predictors of the macrobenthic invertebrate indices than the Piedmont or Southern Appalachian models, using only one variable: topographic complexity. The Southern Appalachian models explained the least amount of the invertebrate variability. The better Southern Appalachian model – for the EPT Biotic Index – comprised two variables that did not appear in any of the other models: rainfall and soil clay content. The amount of rainfall had a positive relationship with better water quality and habitat. Clay had the opposite correlation. The clay relationship seems to make more intuitive sense, because small soil particles can cause a variety of water quality and stream habitat problems. Increasing amounts of rainfall, on the other hand, are often thought to result in water quality

problems, because of increased flow into streams of pollutants such as sediment and excess nutrients following storm events.

The “proportion of riparian forest cover” variable was removed from these analyses because of high correlation (>0.8) with the proportion of forest in the entire watershed. The other variables excluded from the statewide regression analyses, listed above, were also not included in the regional analyses.

3.5 Discussion

The full regression models in this analysis explained between 71 percent and 77 percent of the variability in the macroinvertebrate index scores (Table 3.5). This appears consistent with other analyses of landscape characteristics and water quality. Johnson *et al.* (2001), for example, were able to explain about 75 percent of variability in nitrogen concentration in Pennsylvania from landscape attributes. Herlihy *et al.* (1998) found that land cover explained 58 percent of chloride and 43 percent of nitrate concentration in the Blue Ridge region, and 89 percent chloride and 82 percent nitrate in the Mid-Atlantic Piedmont and Coastal Plain. Basnyat *et al.* (1999) found watershed land use and land cover in Alabama accounted for about 66 percent of variability of nitrate concentration and 76 percent of sediment concentration (although the statistical significance was low: $p=0.31$ and $p=0.5$, respectively). Hunsaker *et al.* (1992) explained 88 percent of water purity in southern Illinois using a variety of landscape variables.

The reduced “best” models – derived using stepwise regression – in the current project explained nearly as much invertebrate variability as the full models, but with only two or three variables. These models could assist in better understanding which landscape

characteristics are most statistically likely to have the greatest impact on stream invertebrate assemblages. Additionally, reducing uncertainty is a key component of ecological risk assessment (Suter 1993), and models with fewer parameters are less likely to contain excessive uncertainty from data collection and variance in the regression model (Draper and Smith 1998). However, these reduced equations might not be useful in determining the impact of site-specific land-use changes on water quality or in comparing the relative importance of a lengthy list of landscape characteristics.

In this study, topographic complexity was the most important variable in all the statewide multiple regression models, indicating that any nonpoint pollution effects on aquatic ecosystems resulting from land use changes are likely to be mitigated or exacerbated by the terrain. Specifically, because greater topographic complexity was related to better stream conditions, steep terrain in some areas may prevent the agricultural and urban development that could result in degraded water quality. Areas with more uniform topography, on the other hand, may be more extensively and permanently altered by human activities. Multiple regression analyses by ecoregion revealed a slightly different result: While topographic complexity was the most important variable affecting macrobenthic community structure in the Piedmont and Coastal Plain (Figure 3.11), it was not a statistically significant variable in the Southern Appalachians. This may reflect the surge in recent decades of home construction in parts of western North Carolina's steep mountainous terrain that were previously considered unfit for such development. Thus, the pollutants associated with development may not come mostly from flatter areas in the mountains, as appears to be the case in the other ecoregions.

The second most important variable was the proportion of watersheds that had been developed. Other researchers have found similar results (Diamond and Serveiss 2001, Sliva and Williams 2001, Wang *et al.* 2001, Basnyat *et al.* 1999, Zandbergen 1998, Snodgrass *et al.* 1997, Wang *et al.* 1997, Lenat and Crawford 1994). This was the case even though developed land cover exhibited a weaker relationship with the benthic indices than forest and agricultural land cover in a series of simple bivariate regression analyses (Table 3.3). The importance of developed land cover clearly increases when its interaction with other landscape characteristics is considered.

Correlations among landscape features and indicators of water quality have been found to vary among ecoregions (Bryce *et al.* 1999, Herlihy *et al.* 1998). We found that the benthic indices are characterized by a different set of significant landscape variables in each region (Table 3.11).

In the Coastal Plain, topographic complexity is the only significant variable, and explains a majority of invertebrate variability. This could indicate that flatter watersheds in the Coastal Plain are more prone to the stream degradation that might accompany the hydrological modifications, such as stream straightening, that are necessary for agricultural, silvicultural, and urban use. Topographic complexity remains the most important variable in the Piedmont, but is joined by developed land cover as a secondary significant factor. The importance of urban cover is not unexpected, since the Piedmont is the state's most densely populated region, and continues to grow. As with the statewide analysis, the Piedmont results may indicate that areas with flatter topography are more likely to have degraded water quality because larger percentages have been converted from forest to other land uses, especially developed. The proportion of developed land cover is the only significant variable

for the North Carolina Biotic Index in the Southern Appalachians, but rainfall and the soil clay content are the important variables for the EPT Biotic Index. These results may indicate a differing sensitivity to types of stream degradation between the two indices – in-stream sediment for the EPT index, and water chemistry and habitat quality for the North Carolina Biotic Index. The EPT Biotic Index, however, is considered a less reliable measure of water quality, especially at high elevation sites and sites at which low numbers of EPT organisms are collected (David Lenat, pers. com.).

3.5.1 Risk Characterization

Although ecological risk assessment has been mainly used to estimate the risks posed by chemicals introduced into the environment (Borouh 1998), it is also considered an important and promising methodology to aid in efforts to control water pollution (Chen *et al.* 1993). Ecological risk assessment is an integrative approach that balances the complexity of scientific analysis with land managers' need for clear and simple answers about the condition of the watershed and the actions needed to achieve certain objectives (Zandbergen 1998).

The results of the current ecological risk assessment indicate that North Carolina watersheds with more uniform topography and with increasing urban development are most at risk for degraded water quality and stream habitat conditions. This appears to be the case in the Piedmont, which is already the most densely populated ecoregion in North Carolina. In the Southern Appalachians, the primary source of risk is development, although other landscape features may also be influential. In the Coastal Plain, urban development is less of a risk factor: While watersheds in the Coastal Plain are “flatter” than the rest of the state, those with more topographic complexity have better water quality. This may be the case in part because flatter areas are more prone to land uses such as agriculture and development,

which were often made possible by hydrologic changes that include the creation of ditches and the straightening of streams.

A useful approach for characterizing the risk of potential land management options is to use the regression equations to “simulate” land use activities, such as change in land cover. The equations allow managers and others to determine the expected existing condition of water quality at any freshwater stream sampling site in North Carolina, by measuring only a few landscape characteristics associated with the site’s catchment. By inserting new variable data in the model equation to reflect proposed landscape changes (such as harvesting a certain percent of forest in the watershed, or requiring a certain proportion of forested cover in riparian zones), the model user could predict the resulting water quality changes. The output of the equation is a score for the site and its catchment, with 0 indicating the presence of stream invertebrates least tolerant of degradation (i.e., better water quality), and 10 indicating an invertebrate community most tolerant of degradation (i.e., lower water quality). This approach could be used throughout the Southeast, with different equations derived for states or ecoregions such as the Southern Appalachians or Piedmont.

The risk assessment vulnerability models for the two indices are listed as the regression equations at the bottom of Tables 3.6 and 3.7; the reduced regression equations are listed in Tables 3.9 and 3.10. To predict as much as 75 percent of the current index score for a given watershed in North Carolina, policymakers and land-use managers would simply have to insert into the equation the appropriate existing topographic complexity, percent developed, and ecoregion location values. To determine the likely impact of certain land-use changes – such as increased development in the watershed – they would simply have to input

the new value into the equation to predict the amount the change resulting from a certain amount of development.

As an example, consider the Swift Creek watershed (Figure 3.3). Located in the Piedmont on the southern edge of the rapidly expanding Raleigh urban area, this is one of several headwater catchments in the Neuse River basin, where riparian protection zones are required along streams because of excess nutrient input into the river system (N.C. Division of Water Quality 2000h). The Swift Creek watershed is 463.89 km², with a watershed shape value of 0.21 and a topographic complexity value of 23 meters (derived from the standard deviation of elevation in a digital elevation model). Only 4.76 percent of the watershed has soil that is at least one-quarter clay. In 1992, the watershed was 24.04 percent agricultural, 63.77 percent forested, and 12.19 percent developed (adjusting out other land uses). That year, it received 20.85 inches of rain during the three months before the benthic macroinvertebrate sample was taken, and 84.26 percent of the area within 30.48 meters (100 feet) of streams was forested.

To predict roughly 77 percent of the value of the North Carolina Biotic Index, the manager or decision-maker would insert the proper values (transformed for normal distribution when necessary, as shown in parentheses following each variable) into the following regression model equation. Values of 0 would be entered for Coastal Plain and Southern Appalachians location because the watershed is located in the Piedmont.

$$\begin{aligned}
 \text{NCBI} = & 6.8303 - 0.5749 [\text{Topographic Complexity (ln)}] + 0.2825 [\text{Percent WS Developed (ln+1)}] \\
 & + 0.8158 [\text{WS Shape (sqrt)}] - 0.006 [\text{Percent WS Forested}] + 0.0023 [\text{Soil Clay Content}] \\
 & + 0.205 [\text{Rain (ln)}] - 0.00004 [\text{Percent Buffer Forested (y}^2)] + 0.0443 [\text{Coastal Plain Location}] \\
 & - 0.0162 [\text{WS Area (ln)}] + 0.0299 [\text{Southern Appalachians Location}] + \epsilon
 \end{aligned}$$

The result is a predicted North Carolina Biotic Index value of 5.8 (+/- 1.33). The actual value of 5.47 is within this range. To simulate an increase in urban cover to from 12.19 percent to 25 percent and a decrease of forest from 63.77 percent to 50 percent, the old values for those variables can be replaced with the new. The result is a predicted North Carolina Biotic Index value of 6.34 (+/- 1.46), which is consistent with the decrease in water quality expected to accompany a significant increase in developed area and a decrease in forest cover. That predicted index value could decrease slightly (indicating possibly improved water quality) to 6.3 (+/- 1.45) if the amount of forest cover within 100 feet of streams increased from 84.26 percent to 90 percent.

Since the regression model explains only 77 percent of the variability in the North Carolina Biotic Index score, the result will not predict an exact score for a given location. It will, however, offer a value that allows decision-makers to ascertain how severe a change in water quality they risk relative to the watershed's original condition, to other potential land-use change scenarios.

This risk characterization approach could be further improved with analyses predicting the amount of land-use change that will occur during a given time frame in the area for which policymakers or land managers are making management and policy decisions. Those values could easily be incorporated into the vulnerability model framework to better assess the possible change in macroinvertebrate index scores, and therefore the expected change in water quality and aquatic ecological integrity.

3.5.2 Uncertainty

Environmental regulatory decisions must be made despite extensive scientific uncertainties (Ruckelshaus 1983). In the ecological risk assessment framework, uncertainty is the “imperfect knowledge concerning the present or future state of the system under consideration; a component of risk resulting from imperfect knowledge of the degree of hazard or of its spatial and temporal pattern of expression” (Suter 1993). Explicitly describing this uncertainty, and quantifying it if possible (Hunsaker *et al.* 1990), is a crucial part of the process, because it allows managers and policy makers to determine the strengths and weaknesses – and, therefore, the overall usefulness – of the results (Reckhow 1994). The best watershed-scale regression model in this study explained nearly 75 percent of the variability in the North Carolina Biotic Index, while the best riparian-scale model predicted 65 percent of the variability in the same metric. Nearly all the regression analyses for this project produced *F*-test results (the ratio between explained and unexplained variance) at $p < 0.0001$, making it unlikely that the resulting regression equations occurred by chance. Model uncertainty could be characterized as the remainder of the variability (about 25 percent) not explained by the models, which is likely to have both stochastic and knowledge uncertainty origins. Model uncertainty may be compounded by the absence of additional variables – such as past land use (Harding *et al.* 1998) or point-source pollution emissions (West 2002) – that might play an important role in determining the state of macrobenthic invertebrate communities in North Carolina.

Uncertainty is associated with parameters as well as with the models they collectively create; in other words, much remains unknown about how exactly landscape variables are related to nonpoint pollution that stresses stream organisms. In our regression analyses, *t*-

tests convey the probability of a random relationship between a site's biotic index scores and associated landscape variables. It is therefore possible to quantify the amount of uncertainty associated with each landscape variable.

The quality of the data used to create these variables may also add to the uncertainty of the results. The stream reach dataset, for example, may not completely reflect the presence and location of intermittent and perennial streams, nor does not include any ephemeral streams, which can contribute significant amounts of sediment and other pollutants to the stream network following storm events (Grey and Henry 2002, Bolton and Ward 1993, Neary *et al.* 1993, Ursic 1991). Similarly, watershed boundaries as delineated may not always reflect the actual boundaries among watersheds. For example, watersheds in the Coastal Plain are often difficult to delineate because of uniformly low topography with little relief, and because many streams have been straightened into ditches that cross pre-existing topographic watershed boundaries. Additionally, the average distance between sampling sites and the nearest weather station was 15.75 kilometers; considerable variability in rainfall can occur within that distance. Finally, several watersheds contained soil associations for which soil clay content was not completely available (Schwarz and Alexander 1995).

3.6 Conclusions

This ecological risk assessment process generated vulnerability model equations that can provide a basis for quantitatively comparing, ranking, and prioritizing risks to water quality, which can be useful in cost-benefit and cost-effectiveness analyses of alternative management options (U.S. Environmental Protection Agency 1998a). Specifically, the

model equations offer a useful approach for characterizing the risk of potential land management options through the “simulation” of land use activities, such as conversion of land cover or implementation of best management practices such as vegetated stream buffers.

There are limits to the value of the empirical approach used to assemble these vulnerability model equations. Such statistical models require large empirical databases and identify correlations with a degree of certainty, but do not generally demonstrate a cause-and-effect relationship. While they are important tools for estimating uncertainties, they have limited value for making predictions across scales of biological organization and for untested stresses (Gentile and Slimak 1992). This assessment, however, was limited to only one scale of biological organization – that of benthic macroinvertebrate communities – and focused on nonpoint source pollution stresses that researchers have long and consistently understood to negatively impact those communities. Additionally, the goal of this research was to predict variability in macroinvertebrate community structure, not to establish specific cause-and-effect relationships, which may not even be possible at the landscape scale.

This project yielded several interesting results about the relationship between landscape characteristics – including land-use – and benthic macroinvertebrate community structure. In general, the results of this study indicate that: (1) Watersheds with more agricultural land cover and developed land cover tend to have benthic macroinvertebrate communities that are more tolerant of stream degradation, which indicates a lower level of aquatic ecological integrity and water quality. (2) Watersheds with more forested land cover tend to have macroinvertebrate communities associated with intolerance for degradation, and therefore with better aquatic ecological integrity and water quality. (3) One land form feature – topographic complexity – and one land-use characteristic – percent developed –

were consistently the most important and most statistically significant variables in explaining macroinvertebrate variability in watershed-wide multiple regression analyses. (4) More research is needed on how these interactions vary by the size of a watershed and the ecoregion in which it is located.

Based on these findings, it appears that water quality and stream ecological integrity – as measured by benthic macroinvertebrate community structure – may be most at risk in North Carolina watersheds with a higher amount of urban development and generally flatter topography.

3.7 Acknowledgements

This paper was prepared as part of a dissertation prepared in partial fulfillment of a master of science degree in natural resources at North Carolina State University. Many thanks are due to committee chairman Frederick W. Cabbage, committee members Gary B. Blank and George R. Hess, and EPA STAR grant principal investigator Rex H. Schaberg.

We thank Dr. Marcia Gumpertz, Amy Nail, and Mark Atlas for their constructive statistics advice, Dr. Jim Gregory for instruction on watershed hydrology and stream ecology, and David Lenat for his insights about the use of the benthic macroinvertebrate indices. This project was supported by the U.S. Environmental Protection Agency through Science to Achieve Results (STAR) grant 2000-STAR-K3. It has not been subjected to EPA review and therefore does not necessarily reflect the views of EPA, and no official endorsement should be inferred.

Table 3.1: Variables used in analysis of stream invertebrate tolerance to stream degradation; the “Benthic Factors” column lists the impacts of the variable on macroinvertebrate tolerance, as described in Chen *et al.* 1993. The table lists the description of each variable included in the analysis, the scale at which it was analyzed (either the entire watershed; the area within 300-, 100-, and 50-foot riparian buffers; or both), the source of the data, the pathways through which it affects macroinvertebrate tolerance, and references to sources for further information on the variable and its relationship to nonpoint pollution.

Variable	Description	Scale	Data Source	Benthic Factors	References
Macroinvertebrate index scores (response)	1) N.C. Biotic Index (NCBI) 2) EPT Biotic Index (Ephemeroptera, Plecoptera, and Trichoptera)	NA	N.C. Division of Water Quality Biological Assessment Unit [1982-2000]	NA	Lenat 1993, Lenat 2002 (pers. comm.), NCDWQ 2001a
Land cover (predictor)	Percent of watershed or buffer that is: 1) Forested 2) Agricultural 3) Developed	watershed and buffer	Multi-Resolution Land Cover (MRLC) GIS raster data [1992]	flow regime, water quality, energy/food source, habitat structure	Basnyat <i>et al.</i> 1999, Diamond and Serveiss 2001, Genito <i>et al.</i> 2002, Lenat and Crawford 1994, Omernick <i>et al.</i> 1981, Sliva and Williams 2001, Sponseller <i>et al.</i> 2001, Wang <i>et al.</i> 1997, 2001
Topographic complexity (predictor)	Standard deviation of elevation of watershed or stream buffer	watershed and buffer	USEPA stream reach GIS data and USGS digital elevation model (DEM)	flow regime, water quality, habitat structure	Phillips 1989, Richards <i>et al.</i> 1996, Riekerk <i>et al.</i> 1988
Mean elevation (predictor)	Mean elevation of watershed or stream buffer	watershed and buffer	USEPA stream reach GIS data and USGS digital elevation model (DEM)	flow regime, water quality	Sponseller <i>et al.</i> 2001
Clay content of soil (predictor)	Percent of watershed or stream buffer with soil having at least 25 percent clay content	watershed and buffer	USGS State Soil and Geographic (STATSGO) GIS database	water quality, habitat structure	Cooper <i>et al.</i> 1987, Richards <i>et al.</i> 1996, Sliva and Williams 2001
Precipitation (predictor)	Amount of rainfall during the three months before stream sample	watershed	NOAA National Weather Service data [1982-2000]	flow regime, water quality	Brooks <i>et al.</i> 1991, Lenat 2002 (pers. comm.), Ward and Elliot 1995
Ecoregion (predictor)	Location in the Coastal Plain, Piedmont, and Southern Appalachians (dummy variable)	watershed	USEPA Ecoregions III GIS data	flow regime, water quality, habitat structure	NCDWQ 2001a Riekerk <i>et al.</i> 1988
Watershed area (predictor)	Watershed area (km ²)	watershed	USEPA stream reach GIS data and USGS digital elevation model (DEM)	flow regime, habitat structure, water quality	Brooks <i>et al.</i> 1991, Richards <i>et al.</i> 1996, Ward and Elliot 1995
Watershed shape (predictor)	Horton's Form Factor (area/square of watershed length)	watershed	USEPA stream reach GIS data and USGS digital elevation model (DEM)	flow regime, water quality	Brooks <i>et al.</i> 1991, Ward and Elliot 1995
Watershed slope/relief ratio (predictor)	Watershed elevation change/ watershed length from outlet to highest point on perimeter	watershed	USEPA stream reach GIS data and USGS digital elevation model (DEM)	flow regime, water quality	Brooks <i>et al.</i> 1991, Riekerk <i>et al.</i> 1988, Sliva and Williams 2001, Ward and Elliot 1995

Table 3.2: Summary of relationship among landscape metrics, biotic indices, and water quality. (Variable transformations are in parentheses).

Landscape Variable	Increasing Value for Landscape Metric Means:		
	NCBI	EPTBI	water quality
Percent Agricultural (sqrt)	↑	↑	↓
Percent Forest	↓	↓	↑
Percent Developed (ln+1)	↑	↑	↓
Topographic Complexity (ln)	↓	↓	↑
Mean Elevation ($y^{0.25}$)	↓	↓	↑
Relief Ratio (ln)	↓	↓	↑
Watershed Area (ln)*	↑	↓	↑
Watershed Shape (sqrt)*	↑	↑	↓ ↑
Rainfall (ln)	↓	↓	↑
Soil Clay Content*	↑	↑	↓

* not statistically significant

Table 3.3: Results of simple regression analysis between land cover variables and both of the benthic macroinvertebrate tolerance indices. (Variable transformations are in parentheses).

	N.C. Biotic Index			EPT Index (ln)		
	r^2	Reg. Coeff.	$pr>F$	r^2	Reg. Coeff.	$pr>F$
Percent Agriculture (sqrt)	0.5101	(+) 0.4128	$p<0.0001$	0.4646	(+) 0.0982	$p<0.0001$
Percent Forest	0.5282	(-) 0.0463	$p<0.0001$	0.4749	(-) 0.0109	$p<0.0001$
Percent Developed (ln+1)	0.0915	(+) 0.4297	$p<0.01$	0.1185	(+) 0.1218	$p<0.005$

Table 3.4: Results of simple regression analysis between land form variables and two benthic macroinvertebrate tolerance indices. (Variable transformations are in parentheses).

	N.C. Biotic Index			EPT Index (ln)		
	r^2	Reg. Coeff.	$pr>F$	r^2	Reg. Coeff.	$pr>F$
Topographic Complexity (ln)	0.6955	(-) 0.6477	<0.0001	0.5976	(-) 0.1496	<0.0001
Mean Elevation ($y^{0.25}$)	0.6538	(-) 0.7038	<0.0001	0.5779	(-) 0.1591	<0.0001
Relief Ratio (ln)	0.6181	(-) 0.5517	<0.0001	0.5083	(-) 0.146	<0.0001
Watershed Area (ln)	0.0001	(+) 0.0092	0.9404*	0.001	(-) 0.0079	0.7962*
Watershed Shape (sqrt)	0.004	(+) 0.4739	0.5777*	0.004	(+) 0.1149	0.588*
Rainfall (ln)	0.1496	(-) 1.2416	<0.001	0.172	(-) 0.3317	<0.0005
Soil Clay Content	0.0053	(+) 0.0023	0.5362*	0.0069	(+) 0.0007	0.482*

* not statistically significant

Table 3.5: Results of regression analysis for full landscape model and each of the two benthic macroinvertebrate tolerance indices

	R^2	$pr > F$	Variables
NCBI	0.7657	<0.0001	10
EPTBI	0.7088	<0.0001	10

Table 3.6: Results of full-model regression analysis for the North Carolina Biotic Index. The regression equation for the model is listed below. (Variable transformations are in parentheses).

R^2	$pr > F$	Intercept	n
0.7657	<0.0001	6.786	74
Landscape Variable	$pr > t$	Reg. Coeff.	Beta Weight
Topographic Complexity (ln)	<0.0001	(-) 0.5749	(-) 0.7403
Percent Watershed Developed (ln+1)	0.0356	(+) 0.2825	(+) 0.1989
Watershed Shape (sqrt)	0.1433	(+) 0.8158	(+) 0.1132
Percent Watershed Forested	0.5912	(-) 0.006	(-) 0.094
Soil Clay Content	0.4433	(+) 0.0023	(+) 0.0708
Rain (ln)	0.4123	(+) 0.205	(+) 0.0639
Percent 100-ft Buffer Forested (y^2)	0.648	(-) 0.00004	(-) 0.0589
Coastal Plain Location (1=yes, 0=no)	0.8574	(+) 0.0443	(+) 0.0177
Watershed Area (ln)	0.8278	(-) 0.0162	(-) 0.0156
So. Appalachians Location (1=yes, 0=no)	0.9393	(+) 0.0299	(+) 0.0127
Regression Equation			
NCBI = 6.8303 - 0.5749 [Topo. Complexity] + 0.2825 [Percent WS Developed] + 0.8158 [WS Shape] - 0.006 [Percent WS Forested] + 0.0023 [Soil Clay Content] + 0.205 [Rain] - 0.00004 [Percent Buffer Forested] + 0.0443 [Coastal Plain Location] - 0.0162 [WS Area] + 0.0299 [Southern Appalachians Location] + e			

Table 3.7: Results of full-model regression analysis for the EPT Biotic Index. The regression equation for the model is listed below. (Variable transformations are in parentheses).

R^2	$pr > F$	Intercept	N
0.7088	<0.0001	1.9922	74
Landscape Variable	$pr > t$	Reg. Coeff.	Beta Weight
Topographic Complexity (ln)	0.0001	(-) 0.1424	(-) 0.7358
Percent Watershed Developed (ln+1)	0.0419	(+) 0.0759	(+) 0.2145
Coastal Plain Location (1=yes, 0=no)	0.09	(-) 0.1176	(+) 0.1884
Percent Watershed Forested	0.6191	(-) 0.0015	(-) 0.097
Watershed Shape (sqrt)	0.2674	(+) 0.1711	(+) 0.0953
So. Appalachians Location (1=yes, 0=no)	0.7935	(-) 0.0286	(-) 0.0485
Percent 100-ft Buffer Forested (y^2)	0.8326	(-) 0.000005	(-) 0.0304
Soil Clay Content	0.8029	(-) 0.0002	(-) 0.0256
Rain (ln)	0.8415	(-) 0.0139	(-) 0.0173
Watershed Area (ln)	0.8715	(-) 0.0034	(-) 0.0129
Regression Equation			
EPTBI = 1.9922 - 0.1424 [Topo. Complexity] + 0.0759 [Percent WS Developed] - 0.1176 [Coastal Plain Location] - 0.0015 [Percent WS Forested] + 0.1711 [WS Shape] - 0.0286 [So. App. Location] - 0.000005 [Percent Buffer Forested] - 0.0002 [Soil Clay Content] - 0.0139 [Rain] - 0.0034 [WS Area] + e			

Table 3.8: Summary of stepwise regression results for the two biotic index models, including the number of variables found to be significant in the partial model, and the percent of the model variability (R^2) explained in the full model that is explained in the partial model.

	R^2	$pr > F$	Variables	% Full Model Variability
NCBI	0.747	<0.0001	2	97.56
EPTBI	0.6958	<0.0001	3	98.17

Table 3.9: Reduced model following stepwise regression for the North Carolina Biotic Index. The regression equation for the model is listed below. (Variable transformations are in parentheses).

R^2	$Pr > F$	Intercept	N
0.747	<0.0001	7.2685	74
Landscape Variable	$pr > t$	Reg. Coeff.	Beta Weight
Topographic Complexity (ln)	<0.0001	(-) 0.6314	(-) 0.8131
Percent Watershed Developed (ln+1)	0.0003	(+) 0.3236	(+) 0.2278
Reduced Regression Equation			
NCBI = 7.2685 - 0.6314 [Topo. Complexity] + 0.3236 [Percent WS Developed]			

Table 3.10: Reduced model following stepwise regression for the EPT Biotic Index. The regression equation for the model is listed below. (Variable transformations are in parentheses).

R^2	$pr > F$	Intercept	N
0.6958	<0.0001	1.9871	74
Landscape Variable	$pr > t$	Reg. Coeff.	Beta Weight
Topographic Complexity (ln)	<0.0001	(-) 0.171	(-) 0.884
Percent Watershed Developed (ln+1)	0.0013	(+) 0.0817	(+) 0.2308
Coastal Plain Location	0.0248	(-) 0.1283	(-) 0.2057
Reduced Regression Equation			
$\text{EPTBI} = 1.8588 - 0.171 [\text{Topo. Complexity}] + 0.0817 [\text{Percent WS Developed}] - 0.1283 [\text{Coastal Plain Location}]$			

Table 3.11: Summary of reduced models, following stepwise regression, for North Carolina's three ecoregions.

Coastal Plain ($n=20$)				
	R^2	$pr>F$	Variables	% Full Model
NCBI	0.6413	<0.0001	Topographic complexity	75.4
EPTBI	0.6576	<0.0001	Topographic complexity	79.6
Piedmont ($n=30$)				
	R^2	$pr>F$	Variables	% Full Model
NCBI	0.505	<0.0001	Topo. complexity; Percent watershed developed	91.3
EPTBI	0.5527	<0.0001	Topo. complexity; Percent watershed developed	95.4
Southern Appalachians ($n=24$)				
	R^2	$pr>F$	Variables	% Full Model
NCBI	0.3292	0.0034	Percent watershed developed	60.6
EPTBI	0.4146	0.0036	Rainfall, Soil clay content	82.7

74 North Carolina Study Watersheds

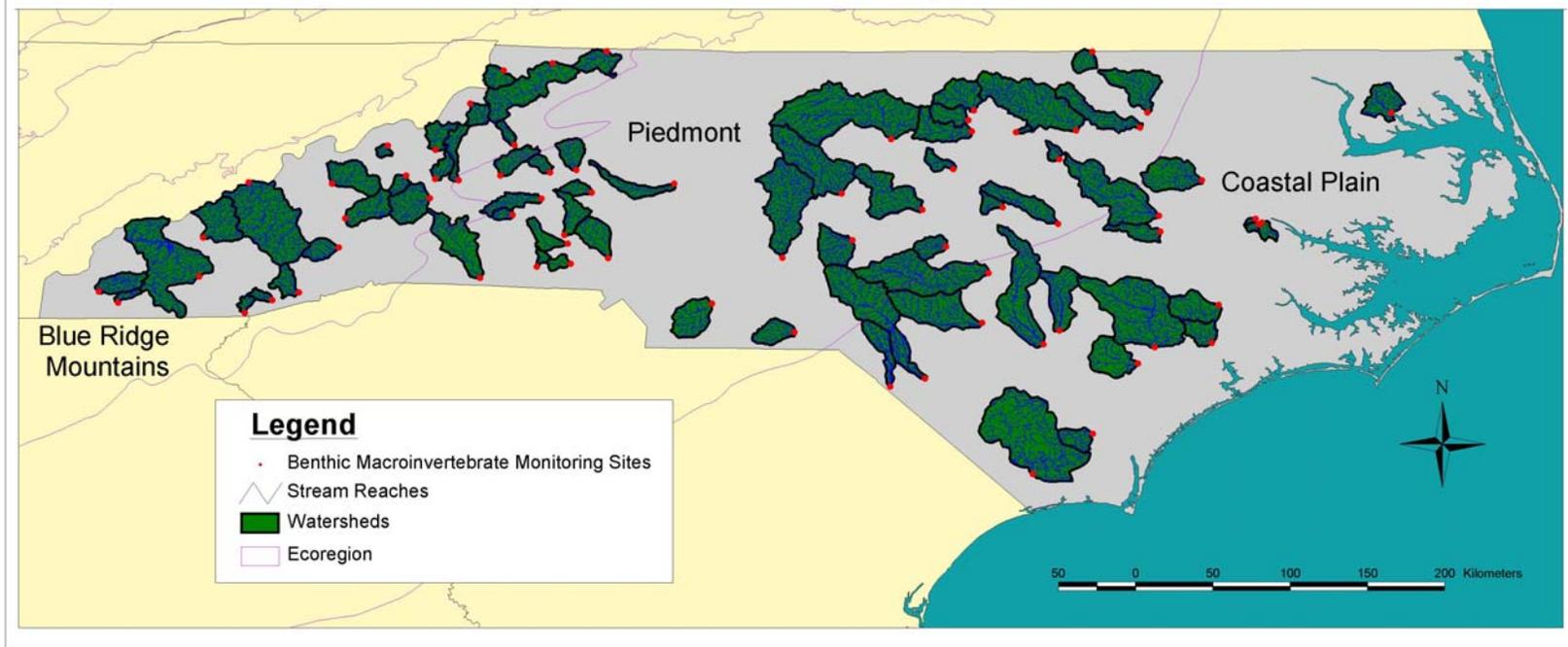


Figure 3.1: The 74 North Carolina study watersheds

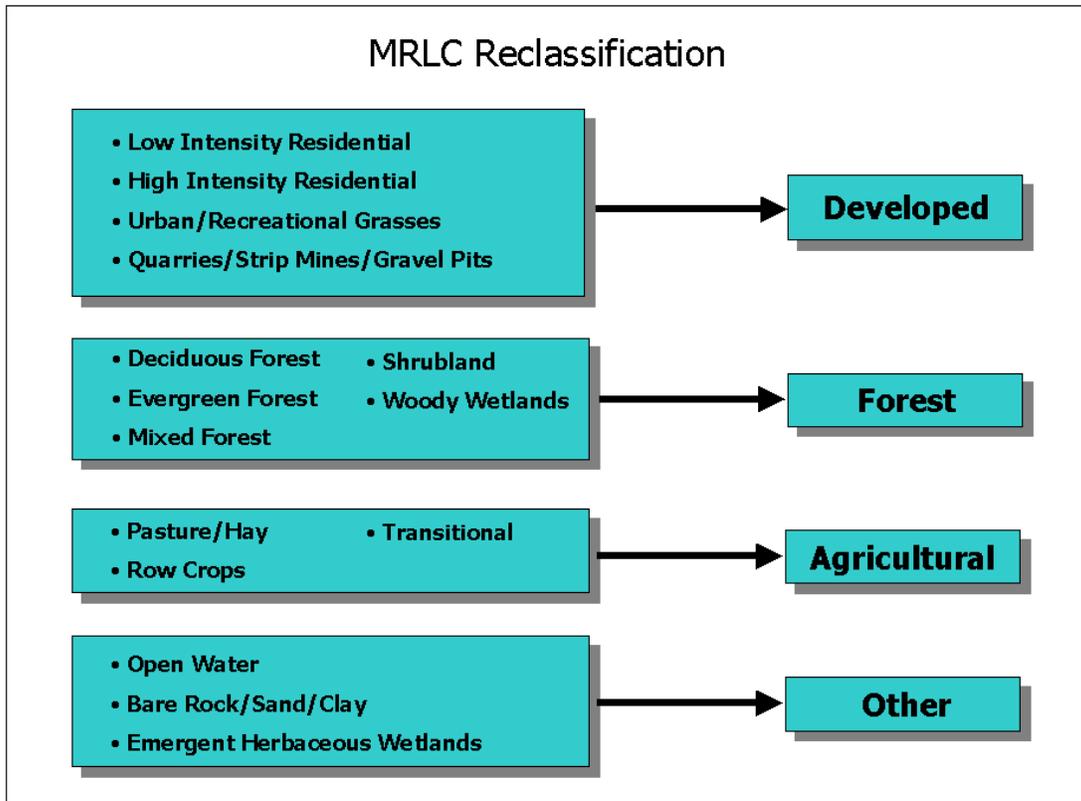


Figure 3.2: Reclassification of 1992 Multi-Resolution Land Characterization (MRLC) National Land Cover Data for North Carolina

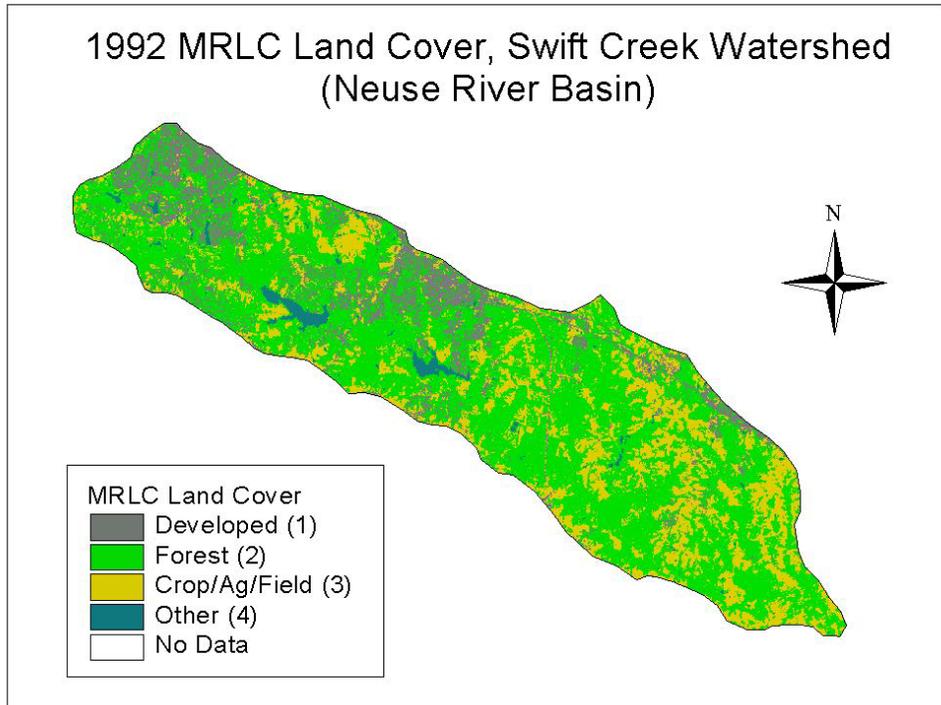


Figure 3.3: 1992 Multi-Resolution Land Characterization land cover for the Swift Creek watershed

Chapter 4

A Matter of Scale: The Best Landscape Predictors of Water Quality Depend on Watershed Size

Prepared for submission to *Journal of the American Water Resources Association*

4.1 Abstract

Researchers disagree about whether riparian zone characteristics or watershed-wide landscape attributes are more significant in determining water quality and the integrity of aquatic biological community structure. The implications of this debate are significant: Further research could shed light on whether the preservation of streamside vegetation adequately mitigates nonpoint source pollution impacts associated with wider land-use change. This study uses stepwise multiple regression analysis of Geographic Information System-derived data to: (1) Determine whether the integrity of North Carolina benthic macroinvertebrate communities is more closely related to landscape characteristics at the scale of riparian zones or entire watersheds; (2) Understand which landscape attributes are associated with aquatic invertebrate communities whose presence indicates degraded stream conditions; (3) Examine whether varying widths of streamside buffers relate differently to the integrity of invertebrate assemblages; and (4) Investigate whether the importance of streamside buffers differs based on a watershed's size. The results indicate that watershed characteristics explain a greater amount of variability in macrobenthic invertebrate community structure (70 to 75 percent) than riparian attributes (58 to 65 percent). While topographic complexity appears to be the most important variable at all scales, the analysis suggests that different land cover characteristics are of secondary importance at different scales: For watersheds, this is developed land cover; at the riparian scale, it is forest cover. This study found only a small difference in the amount of water quality variability explained by riparian buffer zones of two different widths. Finally, the amount of variability in water quality explained by both riparian buffer and watershed-wide forest cover was considerably higher for the smallest quarter of watersheds than the largest quarter.

4.2 Introduction

Between 1988 and 1998, nonpoint pollution sources – especially agriculture and urbanization – were responsible for 70 percent of water pollution in the Southeastern United States, in the form of sedimentation, bacterial pollution, and excess nutrient inputs (West 2002). This pollution presents a threat to the Southeast’s globally significant aquatic biodiversity, including many uncommon and endemic species that inhabit streams and rivers. Of the region’s 176 rare aquatic insect species, for example, 82 percent are found in streams and rivers, as are 88 percent of 165 rare fish species, and 57 percent of 159 rare crustaceans (Herrig and Shute 2002).

Nonpoint source pollution is a landscape-scale phenomenon with a wide variety of origins, including agriculture, silviculture, mining, construction, urban activities, and atmospheric deposition (Neary *et al.* 1988). It is difficult, however, to separate the impacts of these human land uses from such natural features as soil composition and watershed structure. The physical characteristics of streams that shape the biotic communities inhabiting them are influenced by a variety of landscape features, such as geologic attributes, catchment area, and land use (Richards *et al.* 1996). Additionally, our understanding about the ecological processes shaping biodiversity, biotic communities, and watershed conditions is incomplete (Harding *et al.* 1998).

Researchers disagree about whether riparian zone or watershed-wide landscape attributes are more significant in determining water quality and the integrity of stream life. The implications are significant, because the effective long-term management and protection of natural resources requires understanding how the scale of the hazard – in this case,

nonpoint source pollution – affects ecological processes, and determining at what scale the effects of the hazard should be monitored (Hunsaker *et al.* 1990). Additionally, research in this area could help determine whether the preservation of streamside vegetation mitigates the impacts of larger scale land-use change within watersheds.

Riparian vegetation plays a key role in creating and maintaining factors that shape community assemblages of fish, insects, and other organisms: stream temperature, food and energy availability, water chemistry, substrate characteristics, stream system hydrology, and channel morphology (Giller and Malmqvist 1998, Harding *et al.* 1999, Allan 1995).

Removal of riparian vegetation can result in more sediment input into a stream, higher stream temperatures because of increased direct sunlight, and greater nutrient input following the loss of vegetative nutrient uptake (Barbour 1997). Some researchers suggest that nonpoint pollution control programs should focus their efforts on riparian corridors, which make up a small percentage of the entire area of watersheds (*e.g.*, Dodd *et al.* 1994, Large and Petts 1994). For example, a series of Environmental Management Commission regulations in North Carolina restrict activities in the riparian zones of watersheds where water quality has been significantly impacted by nutrients and other pollutants. A 50-foot vegetative buffer is required along streams and other water bodies throughout the Neuse River basin and the Tar-Pamlico River basin (N.C. Division of Water Quality 2000h), and in parts of the Catawba River basin (N.C. Division of Water Quality 2001d). Additionally, at least 11 of the 13 Southeastern states encourage or require the protection of streamside management zones (SMZs) as a best management practice (BMP) during silvicultural activities, although the width of these stream buffers vary by state (Prud'homme and Greis 2002). Differing SMZ widths result, in part, from the lack of sufficient scientific research on the proper riparian

buffer width to prevent the excessive flow of nutrients and other chemical solutes into water bodies (Comerford et al. 1992).

Several researchers have detected significant relationships between water quality and the type of land cover in riparian zones. Working in southeastern Wisconsin, Wang *et al.* (2001) found that connected impervious land cover within a 50-meter buffer along streams, or within a 1.6-kilometer radius upstream of the sampling site, had more influence on fish communities and stream base flow than did comparable amounts of imperviousness further away. Basnyat *et al.* (1999) determined that water quality in watersheds of the Fish River basin of Alabama was highest when passive land uses such as forest and grasslands were located next to streams; nonpassive uses such as agriculture and development had negative effects on water quality. Diamond and Serveiss (2001) found that coal mining had the greatest impact on fish community integrity, followed by the percent of crop and urban land area within 100 meters of streams in the Clinch and Powell river basin of southwestern Virginia. They suggested that protecting and enhancing naturally vegetated riparian corridors, along with better control of mine effluents and urban runoff, could help sustain native fish and mussel populations in the watershed.

Other ecologists, however, have concluded that watershed-wide landscape characteristics are better predictors of water quality than riparian attributes. Conducting research in a southeastern Michigan river basin that was predominantly agricultural but quickly urbanizing, Roth *et al.* (1996) found that stream biotic integrity and habitat quality were negatively correlated with the extent of agriculture and positively correlated with the extent of forest and wetlands. These relationships were strongest at the catchment scale, and became weak and non-significant at local scales; riparian vegetation was a weak secondary

predictor of stream integrity. In southwestern Wisconsin pastureland watersheds, Weigel *et al.* (2000) determined that macroinvertebrate assemblages mostly responded to large-scale watershed influences, such as land cover, rather than riparian conditions. Also working in Wisconsin, Wang *et al.* (1997) found that fish biotic integrity scores and habitat quality scores increased strongly with increasing amounts of forested land, and decreased with increasing amounts of agricultural land. The correlations were significant for 100-meter riparian zones, but were stronger when the entire watershed was considered. Sliva and Williams (2001) concluded that catchment landscape characteristics, including urban land use, slope, and soils, in southern Ontario appeared to have a slightly greater influence on water quality than the composition of a 100-meter riparian zone.

Still other researchers paint a more complex picture of how the landscape affects water quality. According to Sponseller *et al.* (2001), the relative magnitude of the influence of land-use practices may depend on the stream attributes being analyzed. Conducting research in the upper Roanoke River basin of southern Virginia, they found that stream water chemistry was generally related to features at the watershed scale, while stream temperatures and substratum characteristics were strongly influenced by land cover patterns at the riparian corridor and sub-corridor scales. Macroinvertebrate indices, however, were most closely related to riparian corridor cover within 200 meters upstream of sampling sites. Research on macroinvertebrate communities by Richards *et al.* (1996) in central Michigan also produced complicated results. One-hundred-meter stream buffer geology and land-use characteristics were more important than catchment attributes for predicting sediment-related habitat variables, while channel morphology was more strongly associated to catchment-wide conditions. This, the researchers concluded, suggests that whole-catchment characteristics

may be most important for maintaining or restoring stream ecosystems. Meanwhile, Harding *et al.* (1998) concluded that both whole-watershed land use and riparian land use from the 1990s were poor indicators of present-day invertebrate diversity in western North Carolina, compared to watershed land use in the 1950s.

Johnson *et al.* (2001) detected a common theme from this recent research: For larger, higher-order watersheds, land cover proportions alone explain most of the water quality variability, while the spatial pattern of land cover is more important for smaller watersheds, especially headwater stream catchments. As a result, the feasibility of using watershed-wide land cover proportions or spatial pattern measurements for predicting water quality depends on the position of the watershed in the spatial hierarchy of rivers and streams.

Multiple regression analysis of Geographic Information Systems (GIS) spatial data is a common approach for determining the relationship between watershed landscape attributes – especially land use and land cover – and water quality. In an analysis of data from 368 wadeable streams in the mid-Atlantic United States, Herlihy *et al.* (1998) found that concentrations of nutrients, chloride, acid neutralization capacity, and base cations were strongly related to watershed land cover. Higher levels of these stream chemistry variables were correlated with higher proportions of agricultural and urban land cover in watersheds, and with lower proportions of forested cover. Basnyat *et al.* (1999) determined, with regression analysis, that water quality is highest in Alabama's Fish River basin where passive land uses such as forests and grasslands are located adjacent to streams. Johnson *et al.* (2001) regressed two types of pollutant responses on watershed landscape variables in Pennsylvania. Using stepwise regression analysis, Sponseller *et al.* (2002) determined that land-use patterns at five spatial scales explain differences in stream invertebrate assemblages

in the upper Roanoke River basin of southwestern Virginia. Harding *et al.* (1998), also using stepwise regression analysis, compared stream invertebrate and fish diversity with 1950s and 1990s land-use data at several scales from the Little Tennessee and French Broad river basins of western North Carolina.

The current study follows the approach of Diamond and Serveiss (2001) and Hunsaker *et al.* (1992), who both used stepwise regression as part of an ecological risk assessment (U.S. Environmental Protection Agency 1998a) of how land use affects water quality. Diamond and Serveiss examined how human land use and stream habitat quality impact native fish and mussel populations in the Clinch and Powell river basin of southwestern Virginia. Hunsaker *et al.* investigated the relationship between land-use characteristics and water purity, as measured by the electrical conductivity of water.

Employing an ecological risk assessment framework (U.S. Environmental Protection Agency 1998a, Suter 1993), the current research uses stepwise multiple regression analysis of Geographic Information System-derived data to:

- 1) Determine whether indices of North Carolina stream invertebrate community structure is more closely related to landscape characteristics at the scale of riparian zones or entire watersheds;
- 2) Understand which land cover and land form attributes are associated with aquatic invertebrate communities that indicate degraded stream conditions;
- 3) Examine whether varying widths of streamside buffers relate differently to the integrity of invertebrate assemblages; and
- 4) Investigate whether the importance of streamside buffers differs based on a watershed's size.

4.3 Methods

4.3.1 Landscape Variables

For this study, 11 land cover and land form characteristics (Table 4.1) were measured for 74 North Carolina watersheds (Figure 4.1), several at both the watershed scale and for riparian zones 91.44 meters (300 feet) and 30.48 meters (100 feet) on either side of streams. We used ArcView 3.2 (ESRI 1999) to delineate the watersheds and create buffer files, and ArcGIS 8.1 (ESRI 2001) to project all the files to a common coordinate system – North Carolina State Plane, North American Data (NAD) 1983. Of the variables, six were assembled at the watershed-scale and for the two buffer widths (percent forested, percent agricultural, percent developed, topographic complexity, mean elevation, and soil clay content). The remaining five were assembled only at the watershed scale (precipitation, watershed area, watershed shape, relief ratio, and ecoregion).

Three land cover variables were derived at both the watershed and buffer scales – percent forested, percent agricultural, and percent developed – using a raster grid GIS layer from 1992. This information, commissioned by the Multi-Resolution Land Characterization (MRLC) Consortium, is 30-meter Landsat thematic mapper (TM) raster data for the state of North Carolina (Multi-Resolution Land Characterization Consortium 2000). It consists of 15 data classifications, which we reclassified into four: developed, forest, agricultural, and other. For each catchment, the single watershed vector files and the two stream buffer vector files were converted to grids using ArcView 3.2 (ESRI 1999), and were then used to mask the statewide land cover data. For each watershed and buffer, we found the relative proportion of each land cover type by dividing the number of 30-meter cells in a given land cover type

by the total number of cells in the forest, agricultural, and developed classifications. This process eliminated land uses with a classification of “other” from the analysis.

4.3.2 Macroinvertebrate Indices

We determined which of these landscape characteristics have a significant impact on the integrity of stream life, as measured by two metrics that describe the tolerance of benthic macroinvertebrates to stream degradation. These metrics are the North Carolina Biotic Index (NCBI); and an index of Ephemeroptera (mayfly), Plecoptera (stonefly), and Trichoptera (caddisfly) tolerance (Lenat 1993). The NCBI is considered a more reliable indicator of stream chemistry and habitat quality than of in-stream sediment (N.C. Division of Water Quality 2001a), which is among the more important pollutant sources in the Southeast. The EPT Biotic Index (EPTBI) is believed to better measure the impact of sediment in streams (David Lenat, pers. com.), but is considered a less reliable measure of water quality, especially at high elevation sites and sites at which low numbers of EPT organisms are collected. Both indices are scored on a scale of 0 to 10, with 0 indicating the presence of stream invertebrates least tolerant of degradation (and therefore the existence of better water quality), and 10 indicating an invertebrate community most tolerant of degradation (and therefore lower water quality).

For this study, we assembled a complete list of NCDWQ benthic macroinvertebrate sampling sites and scores from across North Carolina, available from the Biological Assessment Unit (N.C. Division of Water Quality 2002a), and from the latest basinwide water quality assessment reports (N.C. Division of Water Quality 1999a, 2000a, 2000b, 2000c, 2000d, 2000e, 2000f, 2001b, 2001c, 2002b). From this list of 2,000 sampling sites,

we selected 74 by excluding those with watersheds extending into other states, those with watersheds larger than 2,500 km², and those without both NCBI and EPT Biotic Index invertebrate samples were taken in about 1992 and about 1997. This final limitation was chosen because our initial goal was to study the relationships between changes in land cover over time (e.g., urban development and deforestation) with changes in water quality tolerance measures. However, adequately compatible land cover data did not exist at these two, or any other, intervals of time.

4.3.3 Data Analyses

We transformed data as needed to achieve a normal distribution for each variable (SAS Institute Inc. 2000). We ran bivariate linear regressions for each of the transformed landscape variables against the invertebrate index variables, to determine the strength and direction between the predictor and response variables. A larger coefficient of determination (r^2) value indicated a stronger relationship between variability in the landscape characteristic and the invertebrate index, while a positive coefficient suggested that a landscape variable is associated with invertebrate taxa tolerant of greater stream degradation. A positive regression coefficient, therefore, indicates the landscape variable has a negative correlation with water quality, while a negative coefficient has a positive effect.

Before running regressions on multiple variables, we tested the multicollinearity of the landscape variables using Pearson correlation analysis (SAS Institute Inc. 2000). When two or more variables were correlated at a level of 0.8 or higher (O'Sullivan and Russel 1999), we removed one or more variables so that only one remained. Relief ratio and average elevation were removed from analyses because both were highly correlated (>0.9)

with topographic complexity, which remained. The proportion of agricultural land cover was almost the reverse of the proportion forested (correlation of -0.98), and was removed from both watershed and buffer multiple regression analyses; the forest variable was included in the analyses. The developed proportion of riparian zones was correlated with the developed proportion of the watershed (>0.8), and was dropped from consideration, as was the agricultural proportion of riparian zones, which was correlated with the proportion of the buffer that is forested. The Southern Appalachian location categorical variable was removed from the buffer analyses because it was correlated with Coastal Plain location and Piedmont location, both of which remained in the analyses.

Stepwise multiple regression (Draper and Smith 1998) was used to analyze the proportion of variability in the stream invertebrate tolerance indices attributable to differences in land cover and land form characteristics. We used an α -level of 0.05 as the significance level for the partial F entry test, and we used the conservative α -level of 0.15 for the partial F removal test (Draper and Smith 1998). We ran multiple regressions in SAS (SAS Institute Inc. 2000) with variables derived at the watershed level, and at the 91.44-meter (300-foot) and 30.48-meter (100-foot) riparian zone widths. A handful of variables – ecoregion, rainfall, watershed area, and watershed shape – were included in both the watershed and riparian regressions.

The resulting coefficient of multiple determination (R^2) for each equation indicates the proportion of variability in the invertebrate tolerance indices attributable to the landscape variables included in the model. The unstandardized regression coefficients for each landscape variable represent that variable's weight and direction in the vulnerability index equation. The standardized (beta weight) regression coefficients indicate the relative

importance of the landscape characteristic relative to the other landscape variables in the model equation.

4.4 Results

4.4.1 Watershed-scale Analysis

The watershed-scale stepwise regression analysis found that only two variables – topographic complexity and percent of watershed developed – explained nearly 75 percent of the variability in the North Carolina Biotic Index ($R^2=0.747$, $n=74$, $p<0.0001$). Topographic complexity was negatively related to invertebrate tolerance to stream degradation – and was therefore associated with better water quality. Meanwhile, watershed development demonstrated a positive relationship with invertebrate tolerance and with degraded stream conditions (Table 4.2). The EPT Biotic Index – measuring the presence of the stream invertebrates most associated with better water quality and stream habitat – required three significant variables to account for almost 70 percent of the variability in the index ($R^2=0.6958$, $n=74$, $p<0.0001$). Those variables were, again, topographic complexity and percent developed, along with whether the sampling site was located in the Coastal Plain (Table 4.3). Since this relationship was positive, sampling sites in the Coastal Plain were associated with stream conditions that were less degraded than in the Southern Appalachians or Piedmont.

4.4.2 Riparian-Scale Analysis

For both of the riparian zone widths, topographic complexity was also a significant landscape variable. The only land cover variable that was statistically significant, however,

was percent forested (negatively associated with degraded stream conditions), not percent developed, which was important at the watershed scale. This may be the case because riparian zones were, on average, more highly forested than watersheds, while watersheds were more heavily developed than the area along streams.

The amount of development in the 74 watersheds varied considerably, from zero to 17.8 percent. The mean percent of development was 3.8 percent. At the riparian zone scale, the average percent of development was 2.9 percent within 300 feet of streams, even though one watershed was 22.7 percent developed within that area. While watersheds and streamside zones were both highly forested, riparian areas were slightly more so, having an average forest cover of 82.2 percent within 300 feet of streams. The watershed with the least amount of forest in its riparian zones had 45.9 percent, while the watershed with the most forest had streamside zones that were entirely forested. Watersheds had an average forest cover 73.1 percent. The watershed with the most forest had 99.9 percent; the one with the least had 36.1 percent.

Stepwise regression analysis at the watershed scale explained nearly three-fourths of the variability in the two invertebrate indices, about 10 percent more variability than accounted for in the streamside zone analyses. The regression results for the two riparian widths (300 feet and 100 feet) were virtually identical, indicating that wider riparian buffer zones might translate into only minor differences in invertebrate community structure. (Only the results associated with the 100-foot streamside zone are reported below and in Tables 4.4 and 4.5). The streamside zone analysis accounted for 65 percent of the variability in the North Carolina Biotic Index ($R^2=0.6517$, $n=74$, $p<0.0001$) and for almost 58 percent of the variability in the EPT Biotic Index ($R^2=0.5756$, $n=74$, $p<0.0001$).

4.4.3 Importance of Topographic Complexity

Topographic complexity – measured by finding the standard deviation of elevation in watershed and stream buffer GIS shapefiles – was the most important landscape variable at both scales. The standardized regression coefficients, which compare the relative importance of the predictor variables in a regression model, were consistently largest for topographic complexity throughout the multiple regression results (Tables 4.2-4.5). This landscape variable was always negatively associated with higher macroinvertebrate tolerance for degraded conditions. In other words, watersheds with higher topographic complexity tend to have less degraded water quality than those that have more uniform topography.

4.4.4 Bivariate Regression Results

The examination of simple bivariate relationships between landscape variables and biotic index results found that watershed characteristics better predicted variability in the indices than did streamside zone characteristics (Table 4.6). For example, watershed topographic complexity yielded an r^2 value of 0.696 ($p < 0.0001$), explaining almost 70 percent of variability in the North Carolina Biotic Index, while the same variable for the 100-foot riparian area explained 62 percent ($r^2 = 0.619$, $p < 0.0001$).

Interestingly, two land cover characteristics for the 300-foot streamside zone were slightly better at predicting variability in biotic indices than for the 100-foot zone (Table 4.7). Percent forest and percent agriculture showed highly similar results, in part because the variables were highly correlated at the scale of streamside areas (as for watersheds). For that

reason, only percent forested was included in the multiple regression analyses at both the riparian and watershed scales.

The soil clay content of watersheds and streamside zones did not have statistically significant simple regression relationships with invertebrate index scores, as was the case for the developed percent of riparian zones. Percent developed was statistically significant at the watershed scale, but accounted for less than 10 percent of the variation in the biotic indices.

The forested composition of streamside zones explained a considerably greater amount of invertebrate variability in the smallest watersheds than in the largest watersheds (Table 4.8), which was expected because riparian areas are believed to have greater influence on the structure and function of smaller streams relative to larger ones (Giller and Malmqvist 1998, Allen 1995). At the same time, watershed forest composition also had a stronger relationship to invertebrate indices in smaller watersheds, which runs contrary to the prevailing thought that land cover proportions alone – and not the distribution of land cover – explains a greater amount of water quality variability with increasingly larger watersheds (Johnson *et al.* 2001).

4.5 Discussion

The results of this study indicate that a considerable amount of the variability in macrobenthic invertebrate community structure can be predicted by landscape characteristics, at both the watershed and of the riparian scale. Watershed attributes, however, explain a greater amount of that variability: 70 to 75 percent, compared to 58 to 65 percent explained by riparian conditions, depending on the biotic index. While topographic complexity appears to be the most important and statistically significant regression variable at all scales, the

analysis suggests that different land cover characteristics are of secondary importance. For watersheds, this is the percent of the area developed; for the streamside zone, it is the percent of forest cover.

The implication of these results appears to be that, while the intactness of forested riparian buffers can mitigate nonpoint pollution impacts on high-quality aquatic biota and water quality in North Carolina, land cover at the larger scale may be able to largely overwhelm these effects. Research by Roth *et al.* (1996) in Michigan produced comparable findings, as did work by Richards *et al.* (1996) in the central part of the same state, and Sliva and Williams (2001) in southern Ontario.

Several factors may account for the greater importance of land cover at the watershed scale than at the riparian scale for the integrity of stream invertebrate communities. First, the stream reach spatial data used in this analysis do not include ephemeral channels, which only contain water immediately following a storm event. These channels are excluded from state and local regulations in North Carolina prohibiting development in riparian corridors, including those that require the maintenance of 50-foot vegetative buffer along streams and other water bodies throughout the Neuse River basin and the Tar-Pamlico River basin (N.C. Division of Water Quality 2000h), and in parts of the Catawba River basin (N.C. Division of Water Quality 2001d). Ephemeral streams, however, are believed to contribute significant amounts of sediment, phosphorus, and other chemicals to the stream network following storm events (Grey and Henry 2002, Bolton and Ward 1993, Neary *et al.* 1993, Ursic 1991).

Second, riparian vegetation is not entirely effective at preventing the flow of pollutants such as sediment (Keim and Shoenholtz 1999) and phosphorus (Comerford *et al.* 1992) into streams, including when practices upslope from the riparian zone produce more

pollutants than can be filtered by the riparian buffer (Nutter and Gaskin 1988, Cooper *et al.* 1987). Additionally, pollutants can bypass riparian vegetation through channelized flow or subsurface pathways (Comerford *et al.* 1992, Omernik *et al.* 1981).

Finally, past land-use practices may continue to have a significant impact on current macrobenthic invertebrate community structure, especially in a state such as North Carolina where agriculture has been widely practiced for longer than a century. After finding that nearly 50 years of riparian reforestation did not return streams in formerly agricultural western North Carolina watersheds to their predisturbance biotic condition, Harding *et al.* (1998) concluded that the preservation of such habitat fragments may not be sufficient to maintain natural biotic diversity in streams. Rather, as they contend, maintenance of such biodiversity may require conservation of much or all of a watershed.

The results of the current analysis partially support the hypothesis expressed by Johnson *et al.* (2001) that the spatial pattern of land cover is increasingly important in explaining water quality variability in watersheds decreasing in size and stream order, while land cover proportions alone are dominant predictors for larger, higher-order watersheds. In our North Carolina study, the amount of variability in the North Carolina Biotic Index explained by the proportion of forest cover in riparian areas was considerably higher and more statistically significant for the smallest quarter of watersheds ($r^2=0.6601$, $n=19$, $p<0.0001$) than for the largest quarter ($r^2=0.0303$, $n=19$, $p=0.4691$). This was expected because riparian vegetation is believed to have a greater influence on water quality of smaller watersheds. At the same time, the watershed-wide land cover proportions were *not* increasingly better predictors of water quality variability in increasingly larger watersheds, as Johnson *et al.* predicted. In fact, the reverse was true: Forest cover was a much better

predictor for the smallest watersheds ($r^2=0.7185$, $n=19$, $p<0.0001$) than for the largest ($r^2=0.4083$, $n=19$, $p<0.005$).

In smaller watersheds, pollutants from agricultural land uses (the proportion of which is highly negatively correlated with percent forest cover) might have to travel less distance, on average, to reach streams. Larger watersheds also have larger-order streams, in which water quality may be less affected by landscape characteristics than by point-source pollution. Ecoregion may also be an influence, since the smallest watersheds were more likely than random to be located in the Southern Appalachians, while the largest were more likely than random to be in the Coastal Plain.

The watershed-scale significance of urban land use, meanwhile, was not unexpected, because many researchers have demonstrated connections between the amount of urban development and water quality. Through direct spatial effects, such as stream channel alternation, increasing amount of impervious surface, and altered water and contaminant transport pathways, urbanization can result in increased stream turbidity, nutrient enrichment, bacterial contamination, organic matter loads, toxic compounds, and temperature (Snodgrass *et al.* 1997). In Alabama, for example, built-up areas were identified as the strongest contributors of nitrogen in one stream pollution model (Basnyat *et al.* 1999). A study of watersheds in the North Carolina Piedmont found low invertebrate taxa richness, very low abundance, and few unique species at the urban site, indicating severe stress (Lenat and Crawford 1994). Research in southern Ontario determined that urban land use had the greatest influence on water quality, and that increasing chemical fluxes to streams accompanied increasing urban land-use density (Sliva and Williams 2001). Forest land use

in that study appeared important in mitigating water quality degradation, which may be the case in North Carolina as well, at least at the riparian scale.

Topographic complexity was the only land form characteristic that was significant at both the watershed and riparian scales, where it was always the most important variable. This negative relationship to invertebrate tolerance (and therefore positive relationship to water quality) was borne out in bivariate regression analyses at both scales. The relationship between topographic complexity and water quality was slightly stronger at the watershed scale than the riparian scale, however. The importance of this variable may be attributable to the less-tillable and developable nature of areas with great topographic complexity, which may be more likely to remain forested. Another factor may be the greater hydrological alteration of stream systems in flatter areas, for agriculture and developed uses. Since the mean elevation of watersheds and buffers exhibited nearly identical bivariate relationships to topographic complexity (and were excluded from analysis because of multicollinearity), it is additionally possible that water quality improves with increasing elevation. In other words, water quality and stream habitat may, as a rule, be better in the Piedmont than in the Coastal Plain, and better in the Southern Appalachians than both.

This may not be surprising, since a wide variety of natural and human-related differences exist among the three regions. For example, the average size of watersheds in the analysis was 748 km² for the Coastal Plain, 447.9 km² for the Piedmont, and 407.2 km² for the Southern Appalachians. The average watershed area having at least 25 percent soil clay content was 16.87 percent for the Coastal Plain, 55.3 percent for the Piedmont, and 11.64 percent in the Southern Appalachians. The mean proportion of development in watersheds

also varied significantly, with 2.2 percent in the Southern Appalachians, 6.17 percent in the Piedmont, and 2.28 percent in the Coastal Plain.

This study found only a small difference in the amount of invertebrate variability explained by riparian buffer zones of differing widths. The percent of forested land cover within the 300-foot streamside zone was slightly better at predicting North Carolina Biotic Index variability ($r^2=0.3407$, $n=74$, $p<0.0001$) than the 100-foot zone ($r^2=0.3004$, $p<0.0001$). This may indicate that a greater amount of intact forest within wider buffers equates with higher-quality invertebrate community structure, and therefore better stream conditions. This is not enough evidence, however, to support policy or management decisions encouraging or requiring wider protected zones for riparian forest, although it may indicate a need for further research in this area.

4.5.1 Policy and Management Implications

The vulnerability model equations that result from this ecological risk assessment process can be used to compare the relative risk to stream biota – and, thus, water quality – from changes in land use coupled with other landscape characteristics. These vulnerability model equations could be a useful tool for policymakers and land managers in the decision-making process.

Additionally, these equations may offer some insight into the relative risks to water quality from land-use activities at the watershed and riparian scales. Specifically, it appears that land-use at the watershed scale is somewhat more important in explaining the variability of stream invertebrate structure than the same characteristics at the riparian scale. More research is clearly needed in this area, but these preliminary results indicate that riparian zone

protection, while important, may not be enough to ensure the integrity of North Carolina's aquatic biota. Following another North Carolina study, Harding *et al.* (1998) concluded that restoration of riparian vegetation may not be sufficient to maintain natural stream biodiversity, especially in streams degraded by long-term human disturbance. Policymakers may have to weigh the costs and benefits of restricting land-use change on a larger scale than the riparian zone, although more research is needed to determine whether an optimal width of riparian preservation exists.

The results of this study and other research like it may assist in the identification, prioritization, and monitoring of areas that appear most susceptible to degraded stream conditions: watersheds with low topographic complexity that are experiencing rapid urbanization. Such areas could be targeted with best management practices, forest protection, vegetation restoration, and other methods for the mitigation of factors – such as increases in surface water flow and erosion potential, reductions in infiltration to groundwater, and higher risk of water quality contamination (Snodgrass *et al.* 1997) – known to cause stream degradation. Differing types of policies and management regimes may be appropriate in different North Carolina ecoregions.

4.6 Acknowledgements

This paper was prepared as part of a dissertation prepared in partial fulfillment of a master of science degree in natural resources at North Carolina State University. Many thanks are due to committee chairman Frederick W. Cabbage, committee members Gary B. Blank and George R. Hess, and EPA STAR grant principal investigator Rex H. Schaberg.

We thank Dr. Marcia Gumpertz, Amy Nail, and Mark Atlas for their constructive statistics advice, Dr. Jim Gregory for instruction on watershed hydrology and stream

ecology, and David Lenat for his insights about the use of the benthic macroinvertebrate indices. Most of the data used in this project were in the public domain, so thanks are in order to the public agencies that made this information available and easily accessible: the U.S. Environmental Protection Agency's Office of Water, the Multi-Resolution Land Characterization Consortium, and the North Carolina Division of Water Quality. This project was supported by the U.S. Environmental Protection Agency through Science to Achieve Results (STAR) grant 2000-STAR-K3. It has not been subjected to EPA review and therefore does not necessarily reflect the views of EPA, and no official endorsement should be inferred.

Table 4.1: Variables used in analysis of stream invertebrate tolerance to stream degradation; the “Benthic Factors” column lists the impacts of the variable on macroinvertebrate tolerance, as described in Chen *et al.* 1993. The table lists the description of each variable included in the analysis, the scale at which it was analyzed (either the entire watershed; the area within 300-, 100-, and 50-foot riparian buffers; or both), the source of the data, the pathways through which it affects macroinvertebrate tolerance, and references to sources for further information on the variable and its relationship to nonpoint pollution.

Variable	Description	Scale	Data Source	Benthic Factors	References
Macroinvertebrate index scores (response)	1) N.C. Biotic Index (NCBI) 2) EPT Biotic Index (Ephemeroptera, Plecoptera, and Trichoptera)	NA	N.C. Division of Water Quality Biological Assessment Unit [1982-2000]	NA	Lenat 1993, Lenat 2002 (pers. comm.), NCDWQ 2001a
Land cover (predictor)	Percent of watershed or buffer that is: 1) Forested 2) Agricultural 3) Developed	watershed and buffer	Multi-Resolution Land Cover (MRLC) GIS raster data [1992]	flow regime, water quality, energy/food source, habitat structure	Basnyat <i>et al.</i> 1999, Diamond and Serveiss 2001, Genito <i>et al.</i> 2002, Lenat and Crawford 1994, Omernick <i>et al.</i> 1981, Sliva and Williams 2001, Sponseller <i>et al.</i> 2001, Wang <i>et al.</i> 1997, 2001
Topographic complexity (predictor)	Standard deviation of elevation of watershed or stream buffer	watershed and buffer	USEPA stream reach GIS data and USGS digital elevation model (DEM)	flow regime, water quality, habitat structure	Phillips 1989, Richards <i>et al.</i> 1996, Riekerk <i>et al.</i> 1988
Mean elevation (predictor)	Mean elevation of watershed or stream buffer	watershed and buffer	USEPA stream reach GIS data and USGS digital elevation model (DEM)	flow regime, water quality	Sponseller <i>et al.</i> 2001
Clay content of soil (predictor)	Percent of watershed or stream buffer with soil having at least 25 percent clay content	watershed and buffer	USGS State Soil and Geographic (STATSGO) GIS database	water quality, habitat structure	Cooper <i>et al.</i> 1987, Richards <i>et al.</i> 1996, Sliva and Williams 2001
Precipitation (predictor)	Amount of rainfall during the three months before stream stream sample	watershed	NOAA National Weather Service data [1982-2000]	flow regime, water quality	Brooks <i>et al.</i> 1991, Lenat 2002 (pers. comm.), Ward and Elliot 1995
Ecoregion (predictor)	Location in the Coastal Plain, Piedmont, and Southern Appalachians (dummy variable)	watershed	USEPA Ecoregions III GIS data	flow regime, water quality, habitat structure	NCDWQ 2001a Riekerk <i>et al.</i> 1988
Watershed area (predictor)	Watershed area (km ²)	watershed	USEPA stream reach GIS data and USGS digital elevation model (DEM)	flow regime, habitat structure, water quality	Brooks <i>et al.</i> 1991, Richards <i>et al.</i> 1996, Ward and Elliot 1995
Watershed shape (predictor)	Horton's Form Factor (area/square of watershed length)	watershed	USEPA stream reach GIS data and USGS digital elevation model (DEM)	flow regime, water quality	Brooks <i>et al.</i> 1991, Ward and Elliot 1995
Watershed slope/relief ratio (predictor)	Watershed elevation change/ watershed length from outlet to highest point on perimeter	watershed	USEPA stream reach GIS data and USGS digital elevation model (DEM)	flow regime, water quality	Brooks <i>et al.</i> 1991, Riekerk <i>et al.</i> 1988, Sliva and Williams 2001, Ward and Elliot 1995

Table 4.2: Stepwise regression results for the North Carolina Biotic Index, watershed scale. The regression equation for the model is listed below. (Variable transformations are in parentheses).

R^2	$Pr > F$	Intercept	n
0.747	<0.0001	7.2685	74
Landscape Variable	$pr > t$	Reg. Coeff.	Beta Weight
Topographic Complexity (ln)	<0.0001	(-) 0.6314	(-) 0.8131
Percent Watershed Developed (ln+1)	0.0003	(+) 0.3236	(+) 0.2278
Regression Equation			
NCBI = 7.2685 - 0.6314 [Topo. Complexity] + 0.3236 [Percent WS Developed]			

Table 4.3: Stepwise regression results for the EPT Biotic Index, watershed scale. The regression equation for the model is listed below. (Variable transformations are in parentheses).

R^2	$pr > F$	Intercept	n
0.6958	<0.0001	1.9871	74
Landscape Variable	$pr > t$	Reg. Coeff.	Beta Weight
Topographic Complexity (ln)	<0.0001	(-) 0.171	(-) 0.884
Percent Watershed Developed (ln+1)	0.0013	(+) 0.0817	(+) 0.2308
Coastal Plain Location	0.0248	(-) 0.1283	(-) 0.2057
Regression Equation			
EPTBI = 1.8588 - 0.171 [Topo. Complexity] + 0.0817 [Percent WS Developed] - 0.1283 [Coastal Plain Location]			

Table 4.4: Stepwise regression results for the North Carolina Biotic Index, 100-foot buffer scale. The regression equation for the model is listed below. (Variable transformations are in parentheses).

R^2	$pr > F$	Intercept	n
0.6517	<0.0001	7.6984	74
Landscape Variable	$pr > t$	Reg. Coeff.	Beta Weight
Topographic Complexity (sqrt)	<0.0001	(-) 0.1684	(-) 0.6812
Percent of Buffer Forested (y^2)	0.0115	(-) 0.00015	(-) 0.21
Reduced Regression Equation			
NCBI = 7.6984 - 0.1684 [Topo. Complexity] - 0.00015 [Percent Buffer Forested]			

Table 4.5: Stepwise regression results for the EPT Biotic Index, 100-foot buffer scale. The regression equation for the model is listed below. (Variable transformations are in parentheses).

R^2	$pr > F$	<i>Intercept</i>	<i>n</i>
0.5756	<0.0001	1.9915	74
<i>Predictor Variable (transformation)</i>	$pr > t$	<i>Reg. Coeff.</i>	<i>Beta Weight</i>
Topographic Complexity (sqrt)	<0.0001	(-) 0.0389	(-) 0.6314
Percent of Buffer Forested (y_2)	0.0217	(-) 0.00004	(-) 0.2098
Regression Equation			
EPTBI = 1.9915 - 0.0389 [Topo. Complexity] - 0.00004 [Percent Buffer Forested]			

Table 4.6: Comparison of simple regressions of land form characteristics at the watershed and 100-foot riparian zone scales, versus biotic index scores. (Transformations of variables are in parentheses).

Landscape Variable	N.C. Biotic Index			EPT Biotic Index		
	r^2	<i>Reg. Coeff.</i>	$pr > F$	r^2	<i>Reg. Coeff.</i>	$pr > F$
<i>Watershed</i>						
Topographic Complexity (ln)	0.6955	(-) 0.6477	<0.0001	0.5976	(-) 0.1496	<0.0001
Mean Elevation ($y^{0.25}$)	0.6538	(-) 0.7038	<0.0001	0.5779	(-) 0.1591	<0.0001
Soil Clay Content	0.0053	(+) 0.0023	0.5362*	0.0069	(+) 0.0007	0.482*
<i>100-Foot Streamside Zone</i>						
Topographic Complexity (ln)	0.6187	(-) 0.1944	p<0.0001	0.5427	(-) 0.0454	p<0.0001
Mean Elevation ($y^{0.25}$)	0.6186	(-) 0.0023	p<0.0001	0.562	(-) 0.0006	p<0.0001
Soil Clay Content	0.0005	(+) 0.0007	0.8525*	0.0019	(+) 0.0003	0.7142*

* not statistically significant

Table 4.7: Comparison of simple regressions of land-use characteristics at the 300-foot, 100-foot, and 50-foot riparian zone scales, versus biotic index scores. (Transformation of variables is in parentheses).

Landscape Variable	N.C. Biotic Index			EPT Biotic Index (ln)		
	r^2	Reg. Coeff.	$pr>F$	r^2	Reg. Coeff.	$pr>F$
<i>Agricultural</i> 300-foot Buffer (sqrt) 100-foot Buffer ($[y+1]^{0.25}$)	0.3273	(+) 0.4796	$p<0.0001$	0.2976	(+) 0.114	$p<0.0001$
	0.2959	(+) 1.8637	$p<0.0001$	0.2711	(+) 0.445	$p<0.0001$
<i>Forest</i> 300-foot Buffer (y^2) 100-foot Buffer (y^2)	0.3407	(-) 0.0004	$p<0.0001$	0.3132	(-) 0.0001	$p<0.0001$
	0.3044	(-) 0.0004	$p<0.0001$	0.2772	(-) 0.0001	$p<0.0001$
<i>Developed</i> 300-foot Buffer (ln+1) 100-foot Buffer (ln+1)	0.007	(+) 0.1243	0.4797*	0.0183	(+) 0.05	0.2508*
	0.0065	(+) 0.1203	0.4942*	0.0169	(+) 0.048	0.2694*

* not statistically significant

Table 4.8: Comparison of simple regressions forested percentage at the watershed scale and 100-foot buffer scale, versus biotic index scores, for the largest and smallest watersheds. (Transformation of variables is in parentheses).

Landscape Variable	N.C. Biotic Index			EPT Biotic Index (ln)		
	r^2	Reg. Coeff.	$pr>F$	r^2	Reg. Coeff.	$pr>F$
<i>Largest Watersheds (n=19)</i> Percent Watershed Forested 100-foot Buffer Forested (y^2)	0.4083	(-) 0.0286	<0.005	0.2409	(-) 0.006	<0.05
	0.0313	(-) 0.0001	0.4691*	0.0216	(-) 0.00002	0.5483*
<i>Smallest Watersheds (n=19)</i> Percent Watershed Forested 100-foot Buffer Forested (y^2)	0.7185	(-) 0.0574	<0.0001	0.6897	(-) 0.0154	<0.0001
	0.6601	(-) 0.0006	<0.0001	0.6822	(-) 0.0002	<0.0001

* not statistically significant

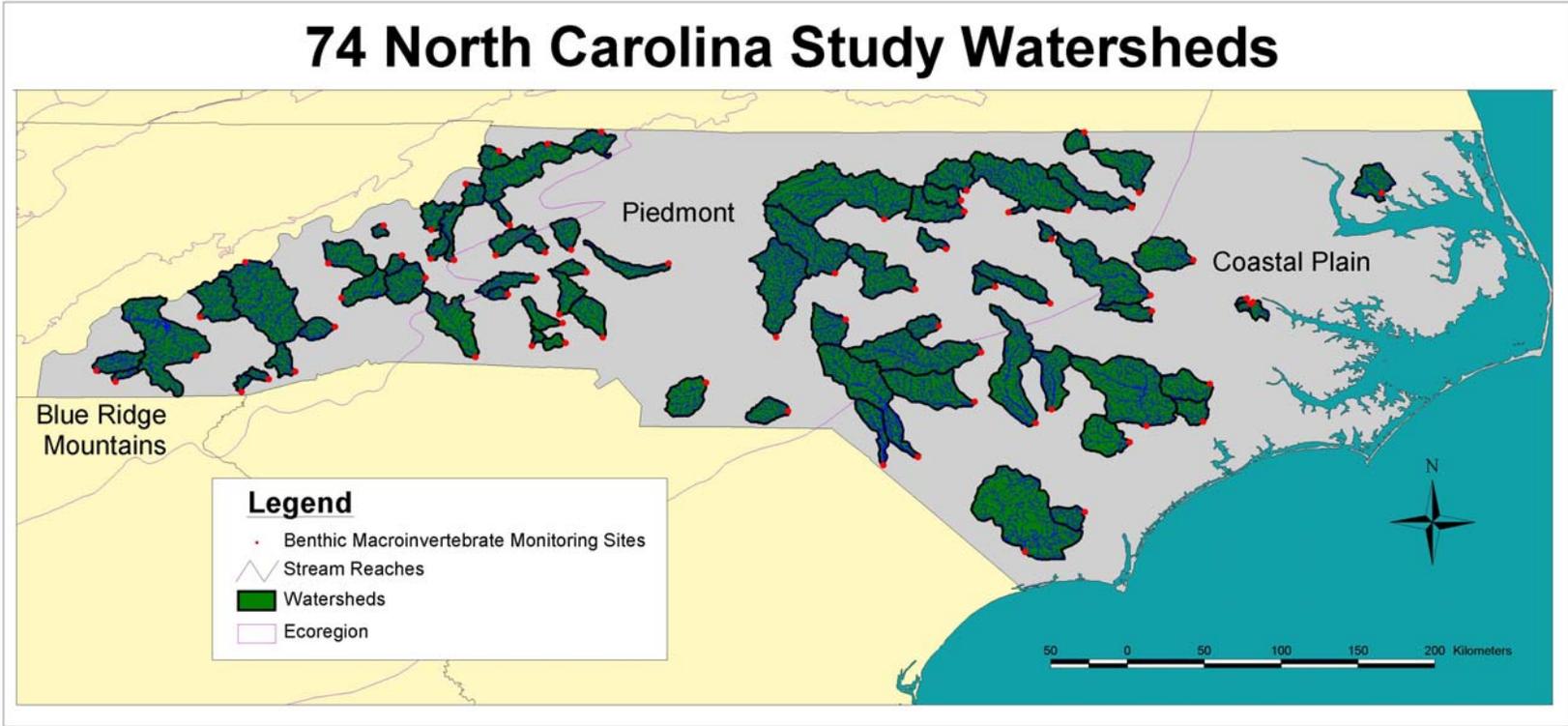


Figure 4.1: The 74 North Carolina study watersheds

Chapter 5

Conclusions, Uncertainties, and Future Work

5.1 Conclusions

This project yielded several interesting findings about the relationship between landscape characteristics – including land-use – and aquatic ecological integrity, as measured by benthic macroinvertebrate community structure. In general, the results of this study appear to indicate that (1) landscape characteristics at the watershed scale better predict variability in benthic macroinvertebrate community structure than characteristics at the riparian scale; (2) land cover variables are of secondary importance to certain land form features, but are still significant predictors of invertebrate community structure; (3) developed land use is the most important land cover variable at the watershed scale, while forested land cover is the most important at the riparian scale; (4) wider riparian buffer zones might translate into only minor differences in invertebrate community structure; and (5) more research is needed on how these interactions vary by the size of a watershed and the ecoregion in which it is located.

Based on these findings, it appears that water quality and stream ecological integrity may be most at risk in North Carolina watersheds where a higher amount of urban development is occurring at the watershed scale, where a lower percentage of forest cover exists in riparian corridors, and where the topography is generally flatter. Conjecture might suggest that silvicultural activities do not have a dramatic nonpoint source pollution impact on water quality and aquatic ecological integrity as long as reforestation – and not urban development or agricultural use – follows harvesting, and as long as timber harvesters leave streamside management zones intact. More research is needed to before drawing such

definitive landscape-scale conclusions about the impacts of silvicultural activities on water quality, however.

Specifically, I found that:

- (1) Watersheds with more agricultural land cover and developed land cover tended to have higher benthic macroinvertebrate index scores. The macroinvertebrates present were more tolerant of stream degradation, which indicates a lower level of aquatic ecological integrity and water quality.
- (2) Watersheds with more forested land cover tended to have lower benthic macroinvertebrate index scores, indicating less tolerance for degradation and therefore better aquatic ecological integrity and water quality.
- (3) The best vulnerability equation generated by regression analysis of 10 landscape characteristics predicted roughly 77 percent of variability in the North Carolina Biotic Index scores (n=74). The best “reduced” model required only two variables to explain 75 percent of the NCBI variability: topographic complexity and percent of watershed developed.
- (4) Using the same data and set of variables, the EPT Biotic Index regression analysis explained almost 71 percent of the variability in the EPT scores. The “reduced” EPT model predicted 70 percent of the variability with the same two variables as the NCBI

- model (topographic complexity and percent watershed developed) along with the ecoregion location of the sampling site.
- (5) One land form feature – topographic complexity – and one land-use characteristic – percent developed – were consistently the most important and most statistically significant variables in explaining macroinvertebrate variability in watershed-wide analyses.
- (6) The type of landscape characteristics significant in affecting index scores varied among North Carolina’s three ecoregions.
- a. For the Coastal Plain (n=20), topographic complexity was the only important and significant variable
 - b. Topographic complexity and percent developed were important in Piedmont watersheds (n=30).
 - c. In the Southern Appalachians (n=24), percent developed was important for the North Carolina Biotic Index, while rainfall and soil clay content were important for the EPT Biotic Index. Topographic complexity may not have been significant, as for all the statewide analyses and for the analyses of the other two ecoregions, because of the recent increase in home construction on steep terrain previously undeveloped.
- (7) Landscape characteristics – including land cover variables – at the watershed scale explained more invertebrate variability than the same variables at the riparian scale

(70 to 75 percent compared to 58 to 65 percent for the EPT Biotic Index and the North Carolina Biotic Index, respectively).

- (8) At the riparian scale, topographic complexity was the most important variable, followed by the forested proportion of the streamside zone. In other words, watersheds with a greater amount of forest cover within riparian zones tend to have lower index scores, and thus higher aquatic ecological integrity and water quality.
- (9) Forested riparian zones of differing widths explained only a small difference in the amount of invertebrate variability. Forest cover in 300-foot riparian zones predicted slightly more variability than forest cover in the 100-foot and 50-foot riparian zones, which were virtually identical.
- (10) The importance of forested cover in riparian zones differed significantly between the largest quarter of watersheds (n=19) and the smallest quarter (n=19). For the smallest watersheds, the relationship between riparian cover and low invertebrate index scores (and therefore better ecological integrity) was quite strong and statistically significant. For the largest watersheds, it was weak and insignificant.

5.1.1 Strengths and Limitations of Analysis

This ecological risk assessment process that produced these results was relatively simple and inexpensive. It took me about six months of intensive work to complete the work. All the data used are free and available to the public. Some of the software used to

complete the project – especially the Geographic Information Systems (ESRI 1999, 2001) and statistical analysis (SAS Institute Inc. 2000) packages are often already available in research settings, but may be considered expensive by some users.

The results are straightforward and generally easy to understand. The vulnerability model equations that resulted from this assessment process can provide a basis for quantitatively comparing, ranking, and prioritizing risks, which can be useful in cost-benefit and cost-effectiveness analyses of alternative management options (U.S. Environmental Protection Agency 1998a). Specifically, they offer a useful approach for characterizing the risk of potential land management options through the “simulation” of land use activities, such as conversion of land cover or implementation of best management practices such as vegetated stream buffers.

There are limits, however, to the value of the empirical approach used to assemble these vulnerability model equations. Such statistical models use large empirical databases and identify correlations with a degree of certainty, but do not generally demonstrate a cause-and-effect relationship. While they are important tools for estimating uncertainties, they have limited value for making predictions across scales of biological organization and for untested stresses (Gentile and Slimak 1992).

This assessment, however, was limited to only one scale of biological organization – that of benthic macroinvertebrate communities – and focused on nonpoint source pollution stresses that researchers have long and consistently understood to negatively impact those communities. Additionally, the goal of this research was to predict variability in macroinvertebrate community structure, not to establish specific cause-and-effect relationships, which may, in fact, not be possible at a landscape scale.

One of the central criticisms of ecological risk assessments is that the endpoints of such analyses are often not of value to society (Renner 1996). Most members of the general public are probably not much concerned about benthic macroinvertebrate community structure. Still, the use of macrobenthic indices is considered an adequate approach to quantifying short-term and long-term degradation of water quality and aquatic habitat structure (N.C. Department of Water Quality 2001a, Adams *et al.* 1995). Additionally, it seems to meet the other requirements for choosing assessment endpoints, as described by Suter (1990).

5.2 Uncertainties

In his second term as administrator of the U.S. Environmental Protection Agency, William D. Ruckelshaus (1983) explained that environmental regulatory decisions must be made despite “enormous scientific uncertainties.” In the ecological risk assessment framework, uncertainty is the “imperfect knowledge concerning the present or future state of the system under consideration; a component of risk resulting from imperfect knowledge of the degree of hazard or of its spatial and temporal pattern of expression” (Suter 1993). Explicitly describing this uncertainty, and quantifying it if possible (Hunsaker *et al.* 1990), is a crucial part of the process, because it allows managers and policymakers to determine the strengths and weaknesses – and, therefore, the overall usefulness – of the results (Reckhow 1994).

Experts in ecological risk assessment divide uncertainty into two categories: knowledge uncertainty and stochastic variability. The first of these results from incomplete understanding or inadequate measurement of the system under study; it can be further

divided into model uncertainty and parameter uncertainty. Meanwhile, the unexplained random variability of the natural environment causes stochastic variability, which is further categorized as spatial or temporal (Hession *et al.* 1996, MacIntosh *et al.* 1994, Suter 1993).

5.2.1 Stochastic Variability

Stochastic variability is a natural property of the system studied (Hession *et al.* 1996). It results from inherent variability of natural populations and ecosystems, and can not be reduced in a dataset with the collection of additional data or through analysis (Hunsaker *et al.* 1990). This is an important issue to consider with the use of biotic indices. Townsend and Riley (1999) caution that site-specific differences in biotic indices may be the result of accidents of geography or history, including the timing of the most recent flood or the vagaries of species recolonization at specific sites. As a result, index results may not vary predictably in response to human impact.

Risk assessors can attempt to reduce stochastic variability to some degree by selecting the proper spatial and temporal scales for the analyses, although such uncertainties may remain large in regional assessments (Hunsaker *et al.* 1990). The current study analyzed the relationship between invertebrate community structure and landscape variables at two scales: the entire watershed, and the zone along streams throughout the watershed. Some uncertainty may have been eliminated by considering a third scale: a subset of the watershed that is a “contributing zone” of possible nonpoint pollutants to a sampling site (Sponseller *et al.* 2001, Basnyat *et al.* 1999). That contributing zone could consist of the riparian area stretching 1 kilometer above the sampling site, or all the area of the watershed within a radius of 1 kilometer upstream from the site.

An additional source of stochastic uncertainty may result from the uneven distribution of sampling sites throughout North Carolina. While the sampling sites and their associated watersheds were roughly evenly divided among North Carolina's three ecoregions (24 in the Southern Appalachians, 30 in the Piedmont, and 20 in the Coastal Plain), the upper Piedmont and the lower Coastal Plain were somewhat underrepresented. This occurred for at least three reasons: 1) the NC Division of Water Quality's repeated sampling sites were not uniformly distributed across the state; 2) watersheds in the upper Piedmont and northern Coastal Plain often contained parts of Virginia, and 3) sites near the mouths of larger rivers such as the Roanoke and the Neuse were not considered because of the size of the watersheds that would have resulted (more than 2,500 km²).

I attempted to minimize temporal stochastic variability by including in my analysis only data from samples that were taken from May to September. Not all samples used in the analysis were taken during the same year, however, because too few samples were taken in 1992, the year for which I had spatial data. I therefore expanded my sample to include sites from which collections had been taken in the summer months from 1990 to 1994.

Long-term data are usually needed to identify significant trends in regional studies (Hunsaker *et al.* 1990). A significant shortcoming of this analysis, and a major contributor to uncertainty, is the limitation of the analysis to only one fairly small window in time. To more adequately understand the relationship between land use and invertebrate community structure, several analyses like this one should be completed at several points in time. Additionally, analysis should be conducted to determine the relationship between invertebrate community change and land cover change, although it may be difficult to separate change in indices over time with data "noise." (See "Future Work" section below.)

5.2.2 Knowledge Uncertainty

The best watershed-scale regression model in this study explained nearly 75 percent of the variability in the North Carolina Biotic Index, while the best riparian-scale model predicted 65 percent of the variability in the same metric. Nearly all the regression analyses for this project produced *F*-test results (the ratio between explained and unexplained variance) at $p < 0.0001$, making it unlikely that the resulting regression equations occurred by chance.

Model uncertainty, therefore, could be characterized as the remainder of the variability (about 25 percent) not explained by the models, which is likely to have both stochastic and knowledge uncertainty origins. I attempted to reduce knowledge uncertainty and the influence of stochastic variability in these models by including only regression variables, through the stepwise process, that were highly statistically significant. As a result, all variables were significant at the $p < 0.05$ level, and most were significant at $p < 0.0001$. Similarly, all the multiple regression models were significant at $p < 0.0001$. Such statistical certainty, however, comes at the risk that meaningful relationships that might not be detected (Karr 1999), especially when they result from the interaction of two or more variables that might not be statistically significant on their own.

Model uncertainty may be compounded by the absence of additional variables – such as past land use (Harding *et al.* 1998) or point-source pollution emissions – that might play an important role in determining aquatic ecological integrity.

Further complicating the picture is the fact that the relationship between the endpoints and indicators used in ecological risk assessment often encompass several levels of

ecological organization. For example, aquatic ecological integrity occurs at the ecosystem level, but biotic indices are measured at the community level. This can introduce important conceptual (model) uncertainties into an ecological risk assessment (Gentile and Slimak 1992).

Uncertainty is associated with parameters as well as with the models they collectively create; in other words, a great deal is unknown about how landscape variables are related to nonpoint pollution stresses to aquatic ecological integrity. Knowledge uncertainty, however, can be reduced by decreasing the possible range of parameter estimates, and by physically sampling the appropriate phenomena – as a result, improving confidence in the estimates of the parameters in the model (Hession et al. 1996).

For each of the regression analyses completed for Chapters 3 and 4, I reported the *t*-test results for each of the regression variables (or parameters). The *t*-test, derived by dividing the regression coefficient by its standard error, conveys the probability of a random relationship between a site's biotic index scores and associated landscape variables. It is therefore possible to see how much uncertainty is associated with each landscape variable. When the result is $p < \sim 0.05$, the relationship is clearly not random. When the result is above about 0.2, the relationship is much more likely to be random, and therefore not statistically significant.

The quality of the data used to create these variables may also be a source of uncertainty, especially since the risk assessment was at the regional level (Hunsaker *et al.* 1990). This may particularly be an issue in the following instances:

- The spatial stream reach data may not completely reflect the presence and location of intermittent and perennial streams, and it does not include any ephemeral streams, which can contribute significant amounts of sediment and other pollutants to the stream network following storm events (Grey and Henry 2002, Bolton and Ward 1993, Neary *et al.* 1993, Ursic 1991).
- The quality of the 1992 MRLC land cover data also may be an issue. A comparison with a 1996 land cover dataset (N.C. Center for Geographic Information and Analysis 1997) revealed significant differences in classifications, although both were Landsat thematic mapper-derived and had similar resolutions (30 meters and 28.5 meters, respectively).
- The watershed boundaries as delineated may not always reflect the actual boundaries among watersheds. Watersheds in the Coastal Plain are particularly difficult to delineate because of uniformly low landscape with little relief, and because many streams have been straightened into ditches that cross pre-existing topographic watershed boundaries.
- The precipitation data used in the analysis was probably the best available without the existence of a network of rain gauges to coincide with each invertebrate sampling site. Still, the average distance between the sampling sites and the nearest weather station was 15.75 kilometers. Considerable variability in rainfall can occur within that distance.

- The GIS soil dataset was often incomplete, so several watersheds contained soil associations for which soil clay content was not completely available (Schwarz and Alexander 1995).

5.3 Future Work

This ecological risk assessment is only a first step in attempting to understand the interactions among land use, nonpoint source pollution, and aquatic ecological integrity. More work is particularly needed on these interactions through time, and on the patterns of land-use change in North Carolina likely to occur in the future.

As Townsend and Riley (1999) note, multi-scale and multi-temporal studies of river function offer the best opportunity to evaluate river health, because of local differences in history and geography, and because of complexities in the interaction of ecosystem responses to human disturbance. Investigating land-use effects through time is also critical for a full understanding and prediction of macroinvertebrate communities in watersheds where land cover is being transformed from forest and agriculture to urban uses (Sponseller *et al.* 2001).

Although inadequate land cover data existed to conduct the current multi-scale analysis at more than one point in time, such research – similar to that of Hunsaker *et al.* (1992) – should be attempted when the proper spatial data become available. For example, a second-generation Multi-Resolution Land Characterization (MRLC) National Land Cover data set for 2000 is currently in production (Multi-Resolution Land Characterization Consortium 2002). This data set may be similar enough to the 1992 data used for the current project (Multi-Resolution Land Characterization Consortium 2000) to offer a useful picture

of land-use change over time, which could then be analyzed in conjunction with North Carolina Division of Water Quality benthic macroinvertebrate monitoring data.

To test this concept, I analyzed the relationship between land cover change over time and the change in benthic macroinvertebrate community structure. This analysis was similar to the single-point-in-time multiple regression analyses described in Chapters 2-4, and used the same 74 invertebrate sampling sites, but employed National Resource Inventory data from 1992 and 1997 (U.S. Department of Agriculture 2002) rather than the 1992 MRLC data. The smallest scale at which NRI land-use data are available is the eight-digit U.S. Geological Survey Hydrologic Unit; these Hydrologic Units are equivalent to river sub-basins that typically are several times larger than the watersheds delineated for the work described in Chapters 2-4 (Figure 5.1).

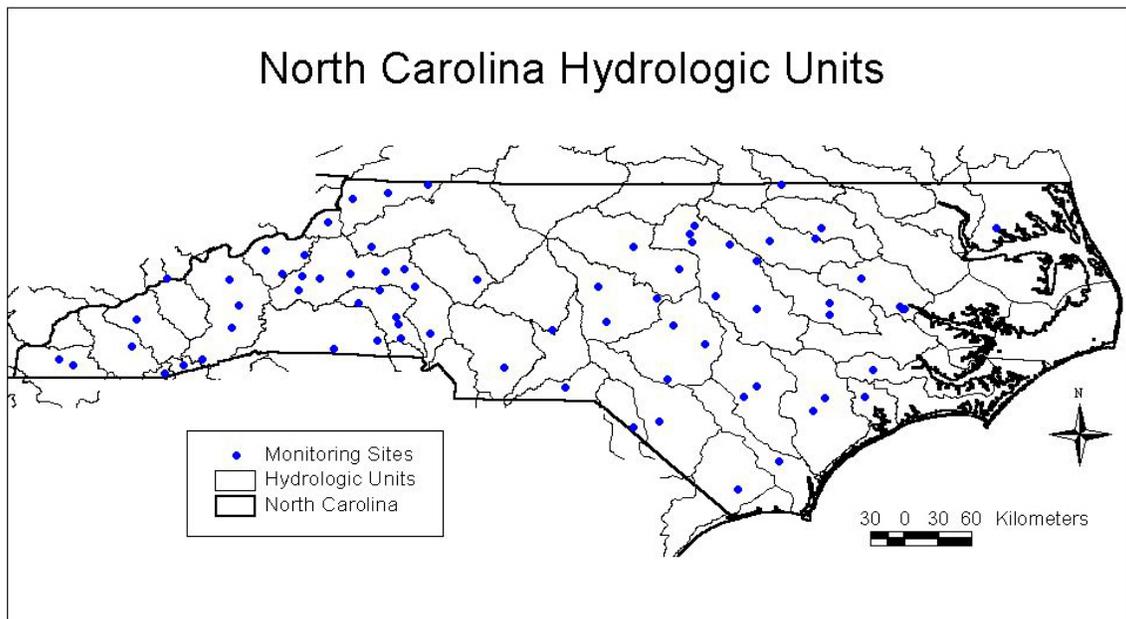


Figure 5.1: U.S. Geological Survey eight-digit Hydrologic Units in North Carolina, and the 74 benthic macrobenthic monitoring sites.

The results of this regression analysis (Table 5.1) indicate that the landscape variables explain about 36 percent of the variability in the change in North Carolina Biotic Index scores. The results are not reliable, however, because of the scale of the NRI land cover data, and because the basins cannot be isolated to macroinvertebrate sample points. The size and direction of the relationships between many of the landscape variables and the biotic index is unexpected and, perhaps, unlikely. For example, all three of the land cover change variables (developed, forest, and agricultural) have positive relationships with increasing invertebrate tolerance for degraded conditions. That would mean increases in all these types of land cover – even forest – are related to worsening water quality. That does not seem possible, but the regression analysis may have been skewed by the scale of the land cover data, by the fact that the change in the index scores were generally toward better water quality and almost always very small, and by the fact that percent of forested and agricultural cover decreased in most basins, while developed cover almost universally increased.

Table 5.1: Results of multiple regression analysis between land form variables and change in the North Carolina Biotic Index between 1992 and 1997. (Variable transformations are in parentheses; the NCBI dependent variable was transformed with an exponential transformation).

<i>R</i> ²	<i>pr</i> > <i>F</i>	<i>Intercept</i>	<i>n</i>
0.3591	0.001	-1.3665	74
<i>Landscape Variable</i>	<i>pr</i> > <i>t</i>	<i>Reg. Coeff.</i>	<i>Beta Weight</i>
Rainfall Change (sqrt)	0.0003	(+) 0.0585	(+) 0.4568
Soil Clay Content	0.0435	(-) 0.0037	(-) 0.2992
Percent Developed Change	0.1848	(+) 0.033	(+) 0.2479
Watershed Area (ln)	0.0147	(+) 0.1198	(+) 0.2975
Percent Forested Change	0.2886	(+) 0.0822	(+) 0.1936
Percent Agricultural Change (exp)	0.5381	(+) 0.076	(+) 0.0865
Coastal Plain Location (1=yes, 0=no)	0.4411	(-) 0.1236	(-) 0.1276
So. Appalachians Location (1=yes, 0=no)	0.946	(+) 0.0159	(+) 0.0173
Watershed Shape (sqrt)	0.8087	(-) 0.0803	(-) 0.0288
Watershed Topographic Complexity (ln)	0.9865	(-) 0.0013	(-) 0.0043

Despite these implausibilities, it appears that a suite of additional landscape variables – including rainfall and soil clay content – may be significant in the results of an analysis like this, and therefore bear more investigation.

Other potential future improvements to this ecological risk assessment include:

- Investigating the use of water quality measures, other than benthic macroinvertebrate community structure, that might better reflect aquatic ecological integrity.
- Using the “transitional” MRLC land-use classification as a surrogate measure for silvicultural activity. These are areas with sparse vegetative cover (less than 25 percent) that are changing from one land cover type to another, often as the result of such land-use activities as timber harvesting.
- Using stream gauge data on stream flow (in the watersheds included in the analyses) rather than relying on precipitation data from sites that could be dozens of kilometers away from the macroinvertebrate sampling site.
- Expanding the land cover categories in the analysis to include pasture and row crop agricultural, low-density and high-density developed, and wetlands, among others.
- Considering alternative approaches to explore the impact of varying riparian buffer widths on water quality and aquatic ecological integrity.

- Analyzing whether riparian zone land cover within differing distances upstream from the invertebrate sampling site (100 meters, 500 meters, 1 kilometer, and 2 kilometers, for example) relates to invertebrate community structure (Diamond and Serveiss 2001, Sponseller *et al.* 2001).
- Investigating the relationship of past land uses, with information perhaps gained from 1940s or 1950s aerial photos, to present-day invertebrate community structure (Harding *et al.* 1998).

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